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DTU Environment
Department of Environmental Engineering

PhD Thesis
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Preface

This PhD thesis is based on research undertaken from August 2007 to January 2011 at the Department of Environmental Engineering, Technical University of Denmark (DTU Environment). The work was carried out under the supervision of Associate Professor Peter Bauer-Gottwein. Roar Jensen of DHI and Flemming Møller of the Danish National Environmental Research Institute (DMU) were co-supervisors. The research was funded by the Danish Research School of Water Resources (FIVA), DHI, and DMU.

The case study area for this research was the Aggitis River basin in northern Greece. It was possible to use this basin as a case study location because of collaboration with DHI. Prior to the start of this project, DHI had developed hydrological models of a number of river basins in northern Greece, including the Aggitis basin, in cooperation with local project partners as part of a planning effort motivated by the European Union Water Framework Directive. This research would not be possible without the support and assistance of DHI and DHI's project partners in Greece, including ENM Consulting Engineers and HPC-Paseco Environmental Consultants in Athens and the Greek Biotope Wetland Center (EKBY) in Thessaloniki.

This research also benefits from an external research stay that was hosted by Professor Manuel Pulido Velazquez at the Technical University of Valencia in Spain.

The content of this thesis is based on three scientific papers. The first has been published in the Journal of Hydrology. The others have been submitted, but not yet accepted.

I Riegels, N., Jensen, R., Bensasson, L., Banou, S., Møller, F., Bauer-Gottwein, P., 2011. Estimating resource costs of compliance with EU WFD ecological status requirements at the river basin scale. Journal of Hydrology, 396 (197-214).

II Riegels, N., Sturm, V., Doulgeris, C., Jensen, R., Møller, F., Bauer-Gottwein, P., 2011. Comparison of two approaches for predicting farmer responses to water price changes in a hydro-economic modelling study. Submitted to Water Resources Research.

III Riegels, N., Pulido Velazquez, M., Sturm, V., Doulgeris, C., Jensen, R., Møller, F., Bauer-Gottwein, P., 2011. Comparison of two water pricing policies in a hydro-economic modelling study. Submitted to the Journal of Water Resources Planning and Management (ASCE).

The papers are referenced using the roman numerals given above after this point.

The papers above are not included in this www-version but can be obtained from the library at DTU Environment. The library can be contacted at the address below:

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Abstract

In regions where water scarcity exists, economic analysis can help identify ways to increase benefits of water use. The European Union's Water Framework Directive (WFD), is an example of a law that requires the use of economic principles, approaches, and instruments in water resources management. One of these instruments is water pricing. This study develops an approach for implementing the water pricing guidelines of the WFD at the river basin scale and then uses hydro-economic modelling to estimate the impacts of applying these guidelines. The central purpose of the WFD is the protection of water resources within the European Union (EU), and water pricing policies are applied with the goal of maximizing economic efficiency while meeting WFD ecological status and groundwater sustainability objectives.

The WFD requires member states to implement water pricing policies that provide incentives for efficient use and contribute to the environmental objectives of the directive. This is interpreted as an endorsement of the use of water pricing as a tool to increase the economic efficiency of water use at the river basin scale. It is assumed that a single river basin planning authority exists and is able to implement a policy that can be applied to all wholesale water users that abstract raw water for economic uses. Water users are assumed to respond to water price changes according to microeconomic theory, either as profit-maximizing producers or utility-maximizing consumers.

This study investigates two water pricing policies. The first is a single volumetric water price that is applied to all wholesale water users in a case study basin. The volumetric price does not vary in time or space and applies to both surface water and groundwater. The second water pricing policy is the same as the first except that surface water and groundwater are priced differently.

Irrigation accounts for almost 90% of total water use in the case study river basin, which is the Aggitis River basin in northern Greece. Because the impacts of water price changes are likely to have a significant impact on irrigation water users, a reasonable model of the economic behavior of irrigation water users is an important element of this study. Two approaches are compared: the residual imputation method and a method based on Positive Mathematical Programming (PMP) that assumes agricultural production can be represented using a functional

form that assumes a constant elasticity of substitution between production factors.

A hydro-economic modelling approach is used to estimate the impact of water pricing on water use. The approach includes a river basin decision support system; methods for predicting water use as a function of water prices; an approach for measuring welfare changes resulting from water price changes; a method for assessing whether environmental flow requirements have been met; an approach for checking groundwater sustainability; an optimization approach that is used to identify appropriate water prices; and an uncertainty analysis approach.

An important conclusion of this study is that water prices would have significant economic impacts on the agriculture sector. These impacts appear to be concentrated on growers of low value crops such as maize, cotton, and fodder crops, which would be unprofitable to grow even at lower water prices.

The PMP and residual imputation approaches predict similar changes in irrigation water use as a function of water price changes. Although the PMP approach has the capacity to predict a wider variety of responses to water price changes, these responses are not observed. Despite the fact that the PMP approach predicts that deficit irrigation will be profitable for many high value crops, the approach does not predict that low value crops will be converted to high value crops as prices increase. Because most irrigated areas are allocated to low value crops in the baseline data set, the result is that land and water use levels predicted by the two approaches are essentially the same. The prediction that high-value irrigated crops will not replace low-value crops is not unreasonable given behavior observed in the baseline data set and highlights the limitations of using economic models calibrated to observed behavior to predict responses to new conditions.

The second water pricing policy, in which surface water and groundwater are priced differently, shifts a small portion of costs imposed by higher water prices from low value crops to high value crops and from small urban/domestic locations to larger locations. Because growers of low value crops will suffer the most from water price increases, the second policy offers the advantage of reducing this burden.

Sammenfatning

I egne af verden med mangel på vand kan økonomisk analyse hjælpe til at øge udbyttet ved vandforbrug. Den Europæiske Unions Vandrammedirektiv er et eksempel på en lov, som anvender økonomiske principper, fremgangsmåder og værktøjer til vandressourceforvaltning. Et af disse værktøjer er prissætning af vand. Med denne afhandling er der udviklet en metode til at implementere Vandrammedirektivets retningslinjer for prissætning på oplandsskala, som herefter er anvendt til hydroøkonomisk modellering for at estimere effekten af disse retningslinjer. Vandrammedirektivets centrale formål er at beskytte vandressourcer inden for den Europæiske Union (EU), og der er anvendt politikker for prissætning af vand med det formål at maksimere den økonomiske effektivitet og samtidig imødekomme Vandrammedirektivets mål for økologisk status og principper for bæredygtighed.

Vandrammedirektivet pålægger medlemslandene at implementere en politik for prissætning af vand, der tilskynder effektivt forbrug og bidrager til at nå de miljømæssige mål, der fastsættes af direktivet. Dette fortolkes som en opfordring til at bruge vandprissætning som et værktøj til at øge den økonomiske effektivitet af vandforbrug på oplandsskala. Det antages, at der eksisterer en myndighed, som kan implementere en politik, der har virkning for alle vanddistributører, som indvinder råvand med et forretningsmæssigt formål. Vandforbrugere antages at reagere på vandpriser svarende til mikroøkonomisk teori, enten som profitmaksimerende producenter eller nyttemaksimerende forbrugere.

Denne afhandling undersøger to politikker for prissætning af vand. Den første er en volumetrisk vandpris, der pålægges alle vanddistributører i et casestudieopland. Den volumetriske vandpris er konstant og gældende for både overfladevand og grundvand. Den anden politik for vandprissætning adskiller sig kun fra den første ved, at overflade- og grundvand er prissat forskelligt.

Kunstvanding udgør næsten 90 % af det totale vandforbrug i casestudiet, Agittis flodens opland i det nordlige Grækenland. Effekten af ændringer i vandprisen forventes derfor at have en markant effekt på brugerne af kunstvanding, hvilket medfører, at en fornuftig model for disses økonomiske adfærd er en vigtig del af denne afhandling. Der sammenlignes to fremgangsmåder: Residual imputation og en metode baseret på positiv matematisk programmering (PMP), som antager,

at landbrugets produktion kan repræsenteres ved en funktion, der antager konstant fleksibilitet mellem produktionsfaktorerne.

Til at estimere effekten af vandprissætning på vandforbruget er der anvendt en hydroøkonomisk model. Fremgangsmåden omfatter et beslutningsstøttesystem for oplandet; metoder til at forudsige vandforbruget som en funktion af prisen på vand; en metode til at måle forskelle i velfærden som følge af ændringer i vandprisen; en metode til at vurdere, om der opstår vandmangel i det omgivende miljø; en metode til at kontrollere bæredygtigheden af grundvandsudnyttelsen; en optimeringsmetode, der bruges til at finde passende vandpriser; samt en metode til analyse af usikkerheden.

En af denne afhandlings vigtige konklusioner er, at prissættelse af vand vil have en markant økonomisk effekt på landbrugssektoren. Denne påvirkning ser ud til at være størst for majs- bomulds- og foderafgrødeavlere, da disse afgrøder vil være urentable selv ved lave vandpriser.

PMP-metoden og residualimputationsmetoden forudsiger enslydende ændringer i vandforbruget ved kunstvanding som følge af ændringer i vandprisen. Selvom PMP-metoden har kapacitet til at forudsige et bredere udsnit af effekterne ved ændring i vandprisen, ses disse effekter ikke. Til trods for at PMP-metoden forudsiger, at en begrænsning af kunstvandingen vil være rentabel for mange højværdiafgrøder, forudsiges ikke, at lavværdiafgrøder vil blive erstattet af højværdiafgrøder, når priserne stiger. Fordi der i baseline datasættet dyrkes lavværdiafgrøder i de fleste kunstvandede områder, bliver de to metoders forudsigelse af arealanvendelse og vandforbrug i alt væsentligt den samme. Forudsigelsen af, at kunstvandede højværdiafgrøder ikke erstatter lavværdiafgrøder, er ikke urimelig på baggrund af den adfærd, der betinger baseline datasættet. Dette understreger begrænsningerne i at bruge økonomiske modeller, der er kalibreret til en given observeret adfærd, til at forudsige reaktionen på nye forhold.

Den anden vandprissætningspolitik med overflade- og grundvand prissat forskelligt flytter en lille del af omkostningerne, skabt ved højere vandpris, fra lavværdiafgrøder til højværdiafgrøder og fra småskala vanddistribution til større skala. Idet dyrkerne af lavværdiafgrøder vil lide mest under en stigning i vandprisen, er fordelen ved denne politik, at denne byrde formindskes.

Contents

1 Introduction

In regions where water scarcity exists, it is possible that existing water management practices do not maximize the value of water to society. In these regions, economic analysis can help identify ways to increase benefits of water use. Even in regions where water is abundant, water management practices may still be sub-optimal because of water quality impacts. However, in this analysis, the focus is on regions confronted by water scarcity and how economic analysis can be applied in these regions to improve water management.

The European Water Framework Directive (WFD), introduced in 2000, outlines requirements for the use of economic principles, approaches, and instruments in water resources management. The central purpose of the WFD is the protection of water resources within the EU, with the goal of achieving good surface water and groundwater status (EU Commission, 2000). In regions where water scarcity exists, the WFD water status requirements imply that EU member states will have to balance ecological and economic water management objectives.

One of the economic principles introduced by the WFD is the principle of recovery of costs of water services. Member states are required to implement water pricing policies that provide for recovery of the costs of providing water services and contribute to the environmental objectives of the WFD. Costs of water services should include what are called “environmental and resource costs”. The assessment of environmental and resource costs has emerged as challenging aspect of WFD implementation (Heinz et al., 2007; Deronzier et al., 2005; Brouwer, 2004; Ward and Pulido-Velazquez, 2009).

A recent EU guidance document regarding environmental and resource costs is an information sheet published by the European working group ECO2 in 2004 (Brouwer). This document defines environmental and resource costs separately. Resource costs are defined as costs arising from depletion of resources beyond natural rates of renewal (e.g., groundwater mining) as well as from economically inefficient allocation of water by existing institutions. Environmental costs are defined as lost environmental benefits, where these benefits are measured using environmental valuation methods such as contingent valuation, travel cost methods, or hedonic pricing (for a recent overview, see Pearce et al., 2006).

Figure 1.1: Polyphytos Reservoir, northern Greece

It appears that the WFD requirement to include environmental and resource costs in water prices is an endorsement of the use of water pricing as a tool to increase the economic efficiency of water use at the river basin scale, with environmental water use values as another economic use of water to consider in the efficiency assessment. In this context, economic efficiency describes the extent to which water use maximizes the value of the water resource to society; an efficient water allocation is assumed to be one in which it is not possible to re-allocate water without reducing overall benefits. If a pricing policy is used as a tool to achieve economic efficiency, then a volumetric water price that is applied consistently to all wholesale water users in a basin could increase efficiency. A single volumetric water price could increase efficiency if wholesale water users respond to water price changes according to microeconomic theory, either as profit-maximizing producers or utility-maximizing consumers. If this assumption holds and all water users are charged the same price, then marginal benefits of water use would be equal among all users. This would mean that it would not be possible to re-allocate water in order to increase benefits of water use.

This analysis assumes that the WFD intends that water pricing policies be designed in order to maximize economic efficiency at the river basin scale. Accordingly, the analysis assumes that it is necessary to price water at the wholesale level. It is assumed that a single river basin planning authority exists and is able to implement a policy that can be applied to all wholesale water users that abstract raw water for human use. The policy describes prices that are charged to wholesale water users in the basin including irrigation districts, municipalities, large industries, and other large entities that abstract raw water.

This study investigates the use of two water pricing policies. The first is a single uniform volumetric water price that is applied to all wholesale water users in a case study river basin. The volumetric price does not vary in time or space and applies to both surface water and groundwater. The second water pricing policy is the same as the first except that surface water and groundwater are priced differently. Surface water is priced at a uniform price, while groundwater is priced by using the price of energy as a surrogate for a water price.

Although a consistent water price appears to meet the criterion of economic efficiency, it may be possible to increase the total benefit of water use through the use of water prices that vary in time and space. Water prices are only useful for improving economic efficiency if water scarcity exists; otherwise, the economic value of water would be maximized by allowing all users to use water for free. Because water supply varies in time and space, prices that are useful for controlling water in use at some times of year or in some parts of a river basin may not be appropriate at other times or year or in other parts of the basin. Therefore, the pricing policies investigated in this study most likely do not maximize the economic value of water.

Water prices are not used to identify efficient levels of use for two categories of water use in this study. These categories are environmental water use and future groundwater use. Although the WFD calls for environmental water use valuation, this analysis does not attempt to value environmental flows or include these values in measurements of economic efficiency. Instead, WFD environmental flow objectives are implemented as a constraint. In addition, this analysis does not investigate whether there is an economically efficient rate at which groundwater can be mined. Instead, groundwater use is constrained so that use does not exceed flows that can be captured from river base flows. This approach is consistent with the WFD, which establishes environmental flow standards and prohibits groundwater mining. Indeed, the WFD objective of economic efficiency should be interpreted as a requirement for economic efficiency subject to environmental and groundwater sustainability constraints.

The author is not aware of any comprehensive investigation in Europe of how a consistent water pricing policy could be implemented across water use categories at the river basin scale, although this is what the WFD appears to require. In southern Europe, the location of the case study river basin, any basin-wide water pricing policy must give careful consideration to the agriculture sector. In some regions of southern Europe, irrigation water use accounts for as much as 80% of total use and can be a principal driver of economic activity (European Environment Agency, 2009). In the case study basin, the Aggitis River basin in northern Greece, irrigation accounts for almost 90% of total water use.

Because the impacts of water price changes are likely to affect irrigation before other water uses, a reasonable model of the economic behavior of irrigation water users is an important element of this kind of study. In this study, two approaches are compared: the residual imputation method and a method based on Positive Mathematical Programming that assumes agricultural production can be represented using a functional form that assumes a constant elasticity of substitution between production factors.

A hydro-economic modelling approach is used to estimate the impact of implementing water pricing at the basin scale. In a hydro-economic modelling approach, a hydrological model is linked with economic tools that measure the economic values of water uses defined within the model. A detailed overview of the hydro-economic modelling approach is provided in the Methods section.

2 Literature review

In the first part of this review, an overview of the field of hydro-economic modelling is provided. The second part of this review provides an overview of the European Union Water Framework Directive (WFD) with a focus on the economic aspects of the WFD. Because of the importance of irrigation water use in this study, literature relating to the economics of irrigation water use is also reviewed.

2.1 Hydro-economic modelling

Hydro-economic modelling is a term used to describe water resource modelling studies that value some or all water uses using a common metric, usually monetary units, to compare values across time and space and among different water use categories. The central concept is that anthropogenic water uses should not be represented as fixed requirements in water resources planning models but rather as demand functions that are based on economic uses of water. Hydro-economic modelling often involves the use of optimization approaches to identify ways to increase the economic efficiency of water use or other social goals. Harou et al. (2009), have recently completed a comprehensive overview of the field. Cai (2008) and Brouwer and Hofkes (2008) have also completed recent methodological papers.

In the recent hydro-economic modelling literature, three efforts stand out. The first includes studies associated with the development and application of the CALVIN model in California (Draper et al., 2003; Jenkins et al., 2004; Pulido-Velazquez et al., 2004; Medellin-Azuara et al., 2007; Harou and Lund, 2008). The second is the work led by Cai on the Maipo River basin in Chile (Cai and Rosegrant, 2004; Cai et al., 2006; Cai and Wang, 2006; Cai et al., 2008; Cai, 2008). The third outstanding body of work is that of Pulido-Velazquez and collaborators in Spain (Pulido-Velazquez et al., 2006; Pulido-Velazquez et al., 2008) and New Mexico (Ward and Pulido-Velazquez, 2008; Ward and Pulido-Velazquez, 2009).

The CALVIN model ((Draper et al., 2003; Jenkins et al., 2004) is the product of an ambitious effort to model the California water system using a hydro-economic approach. California has a highly developed water supply infrastructure that includes a large network of reservoirs and conveyance facilities. The system was developed to move water in time and space, from the winter in the north part of the state, where precipitation is abundant, to the summer in the south part of the state, where water use is highest. Because precipitation is highly variable from year to year, the system also transfers water from wet years to dry years through multi-year storage facilities. The CALVIN model is a network flow optimization model in which urban, agricultural, and industry demands are represented using economic benefit functions. The objective function in the model maximizes the economic value of water use at all locations over a 72-year simulation period that runs on a monthly time step. The 72-year period uses hydrological data from 1922-1993 in order to simulate a range of water conditions that may be expected in the future. Economic benefit functions are based on projected uses of water for the year 2020, and the optimization problem is constrained by environmental flow requirements, reservoir and conveyance capacity limits, and flood control operations.

Perhaps the central innovation of the CALVIN effort is the interpretation of timeseries of shadow prices on constraints and marginal water values at water demand location as indicators of water use values. Draper et al. (2003) and Jenkins et al. (2004) provide averages of these timeseries values in order to compare relative values of water in different parts of the state, benefits associated with infrastructure improvements, and opportunity costs of environmental flow requirements. Model results suggest that removal of existing water allocation policies, which in the CALVIN optimization framework is tantamount to the introduction of water markets, can increase benefits and reduce opportunity costs of environmental requirements throughout the state. Shadow prices on conveyance capacity constraints are also presented as indicators of where expanded conveyance can increase system flexibility. When this approach is used with the system constrained to baseline operating rules and allocation policies, it is found that marginal values of urban water uses are 1-2 orders of magnitude higher than marginal values of irrigation water uses. Some limitations of the CALVIN approach include the assumption of perfect foresight in the optimization of water use over the 72-year hydrological record and the exclusion of hydropower values from the water valuation framework.

In addition to statewide results presented in Draper et al. (2003) and Jenkins et al. (2004), the CALVIN model and approach are used in more detailed studies of regional water issues in California. These include studies of conjunctive use and water banking in southern California (Pulido-Velazquez et al., 2004) and strategies for ending groundwater overdraft in the Tulare Basin of central California (Harou and Lund, 2008). These studies also use timeseries values of shadow prices and marginal values as indicators of benefits to be gained from more flexible operations and infrastructure expansion. The CALVIN model is extended to Baja California in Mexico by Medellin-Azuara et al. (2007), who use the approach to estimate opportunity costs of allocating more water to environmental purposes.

The most comprehensive hydro-economic modelling effort to date is the work of Cai and collaborators on the Maipo River basin in Chile. Although the Maipo basin is less complex than the California statewide system, the representation developed by Cai and others is considerably more detailed than the CALVIN representation. Most of the effort is documented in Cai et al. (2006). The Maipo representation includes elements that simulate basin hydrology; water storage and conveyance infrastructure; a water quality element to simulate the salt balance; agronomic components that simulate crop yield and the soil water balance; economic components that estimate water use values for agricultural, urban/industrial, and hydropower water uses; and methods for simulating water allocation institutions including water rights, a water market, and basin-scale water pricing.

An interesting feature of the Maipo basin modelling effort is the detailed representation of agricultural water use. Crop yield functions are developed for all crop types in the study area that estimate yield as a function of water quantity, salinity, and irrigation technology. The multi-input production functions facilitate the investigation of relationships between field-level irrigation efficiency, basin-level irrigation efficiency, and economic efficiency. It is sometimes proposed that improvements in irrigation technology can result in water savings. Although this is true at the field level, Cai et al. (2003) demonstrate that improvements in field-level irrigation efficiency do not necessarily result in improvements in basin-level irrigation efficiency because field-level efficiency improvements reduce return flows that are available for downstream use. The relationship between field-level irrigation efficiency and economic efficiency can also be counter-intuitive. For example, Cai et al. (2003) find that when water is allocated according the objective of basin-level economic efficiency, field-level irrigation efficiency decreases for some crops. Because of salt-leaching requirements, maximum yields occur when water use exceeds crop evapotranspiration demand. For some high value crops, it is profit-maximizing to use excess water because the benefit resulting from increased yield exceeds the opportunity cost of water use. Cai et al. (2003) also find that allocating water to maximize basin-scale economic efficiency causes water use to shift from downstream to upstream, presumably because elevated salinity levels in downstream flows increase the amount of water that must be used for salt-leaching.

Cai and Wang (2006) use the Maipo River basin case to demonstrate an innovative approach to calibrating a hydro-economic model. The Maipo River basin model is similar to the CALVIN model in that it optimizes water use over a sequence of hydrological input data using perfect foresight. In this situation, it is difficult to recreate observed water use levels without constraining the model to a point that makes it less useful for analysis of alternative scenarios. Cai and Wang (2006) develop an approach based on Positive Mathematical Programming (Howitt, 1995a) that estimates a number of parameters used to calibrate the model. Among these are penalty costs associated with observed land and water use in the irrigation sector that constrain the model to reproduce observed land and water use under optimization with perfect foresight. These penalty costs are then included when the model is used to evaluate alternative scenarios such as water pricing. The inclusion of penalty costs in the model is justified on the grounds that these costs represent physical processes and/or socioeconomic conditions that drive real-world operations but are not represented in the model framework to due to lack of data or understanding.

Pulido-Velazquez et al. (2006; 2008) apply an approach similar to the approach used in the CALVIN model to the Adra River basin in Spain. The major innovation is a detailed simulation of groundwater flow processes, including estimates of changes in well heads. The well head estimates are used to constrain groundwater pumping in order to limit saltwater intrusion. As in the CALVIN model, timeseries estimates of shadow prices on reservoir capacity levels, minimum flow requirements, and well head constraints are interpreted as estimates of opportunity costs of these constraints.

Ward and Pulido-Velazquez (2009) apply a similar approach to the Rio Grande basin in New Mexico and use the approach to estimate the impact of water pricing at the basin scale. The modelling approach is used to estimate the impact of using a two-tiered pricing structure in which water required for household subsistence is priced cheaply while other uses are priced according to marginal costs of water supply. All users are assumed to pay for water at a price equal to the marginal cost of supply, and transfers are allowed in order to maximize basin economic efficiency. In addition to urban/domestic and irrigation water use values, the water valuation approach also includes an estimate of water values associated with sport fishing at reservoirs in the case study basin. The modelling approach is innovative because of the two-tiered pricing policy for urban/domestic users and the inclusion of environmental water use values related to recreational benefits.

Uncertainty analysis appears to be an under-developed aspect of hydro-economic modelling. Uncertainty analysis can be used in hydro-economic modelling to estimate the impact of uncertain input data and model structures on model predictions. In a recent overview of hydro-economic modelling, Harou et al. (2009) recommend the use of sensitivity analyses to reveal boundary conditions, parameters, or model components with significant impacts on model results. However, they point out that sensitivity analyses that evaluate one parameter or model structure at a time may not identify impacts arising from simultaneous variation of multiple parameters. Cai (2008) also recommends the use of sensitivity analyses. Cai further suggests that model calibration can be used to identify reasonable values of uncertain economic and hydrologic parameters (e.g., Cai and Wang, 2006). Both Harou et al. (2009) and Cai (2008) recommend that stochastic optimization methods be applied to hydro-economic modelling, but note that computational challenges have made progress difficult.

2.2 The European Union Water Framework Directive

The economic analysis requirements of the WFD would seem to require a hydro-economic modelling approach because of the requirement to analyze the economic efficiency of water use at the river basin scale. Heinz et al. (2007) have written a methodological paper with recommendations on how to apply hydro-economic modelling in the context of the WFD. The paper recommends hydro-economic modelling a tool that can help identify physical and institutional constraints that result in economic inefficiency.

A number of researchers and practitioners have developed approaches for using hydro-economic modelling to address the economic analysis requirements of the WFD. Pulido-Velazquez et al (2006; 2008) developed a model of the Adra River basin in Spain and optimized the economic value of water use over a ten-year sequence of historical flows to identify how institutional and infrastructure constraints limit economic efficiency. Volk et al. (2008) used a hydro-economic approach to estimate opportunity costs to agriculture of meeting WFD water quality requirements in the Ems River basin in Germany. Bateman et al. (2006) outlined an ambitious program to estimate opportunity costs and benefits of meeting WFD water quality requirements that included a major effort to measure environmental water use values using stated preference techniques. Ward and Pulido-Velazquez (2009) measured opportunity costs of existing water allocation policies at the basin scale by comparing overall net benefits of water use to an optimized case; the water valuation framework used included estimates of irrigation, urban/domestic, and recreational water use values.

2.3 Modelling the economic behavior of irrigation water users

Because the importance of the irrigation sector, particularly in semi-arid regions, methods for modelling the economic behavior of irrigation water users are an important element of any hydro-economic modelling study. It is frequently assumed that irrigation water users can be modeled as profit-maximizing producers. Under this assumption, agricultural production is described using a production function, where output is a function of factor input levels. The irrigation water user is then assumed to optimize with the goal of profit maximization subject to input and output prices. Young (2005) has published an excellent reference describing various methods for modelling farmer behavior under the assumption of profit maximization. McKinney et al. (1999) have also published an overview of methods for developing production functions that can be used to describe the economic behavior of farmers.

Although the assumption of profit maximization appears to be the most common approach to modelling the economic behavior of irrigation water users, other approaches have been developed. Multi-criteria Decision Making (MCDM) has been applied in Spain (e.g., Gomez-Limon and Berbel, 2000; Gomez-Limon and Riesgo, 2004) and Greece (e.g., Latinopoulos, 2008) to predict how farmers might respond to water price changes imposed as part of the WFD. These and other studies were motivated by observations that other criteria besides profit maximization may affect the economic behavior of farmers. In the MCDM approach, factors such as risk minimization and complexity minimization are included along with profit maximization to model farmers' decision-making. A general conclusion of these studies is that costs imposed by volumetric water pricing would place an excessive burden on the agriculture sector. The MCDM approach is more data-intensive than some other approaches because observations from a number of different years are needed to estimate a set of weights that are used to measure the relative importance of factors motivating farmers' decision-making.

It is interesting to note that the MCDM studies cited here have arrived at different conclusions regarding the utility of deficit irrigation as a response to water price changes. Deficit irrigation is a strategy in which water is applied to crops at a rate that is not sufficient for evapotranspiration at the potential rate. Although this strategy reduces yields, it can be profit maximizing when water costs are high, particularly if yield reductions are small relative to water use reductions (English and Raja, 1996). Latinopoulos (2008) concludes that farmers would use deficit irrigation at higher water price changes, while Gomez-Limon and Riesgo (2004) maintain that deficit irrigation is not a realistic option for famers in their case study area. The differing conclusions regarding the utility of deficit irrigation can perhaps be explained by the crop types that were considered in the two studies. Gomez-Limon and Riesgo (2004) considered low-value crops such as maize and sugar beet, while Latinopoulos (2008) considered a mixture of high-value and low-value crops. It may be that only high-value crops remain profitable when deficit irrigation is used. This hypothesis is supported by Lorite et al. (2007), who argue that shifting land to high-value crops will increase net income when water scarcity is present. There is evidence that deficit irrigation can be used successfully in the production of vine and tree crops (e.g., Fereres and Soriano, 2007; Ruiz-Sanchez et al., 2010).

2.4 Irrigation water pricing

Johansson (2000) has provided a review of the irrigation water pricing literature. The review groups irrigation water pricing methods into three main categories: volumetric methods, non-volumetric methods, and market-based methods. According to microeconomic theory, volumetric water pricing can maximize the economic efficiency of water use if prices are set equal to marginal costs of supply. The review highlights a number of factors that limit the extent to which volumetric pricing can achieve this goal in practice. Probably the most significant obstacle is the difficulty of quantifying all marginal costs of water supply, which should include both delivery costs and opportunity costs. Water supply and demand vary in space and time, making opportunity costs more difficult to measure. Some water-related goods such as flood control and environmental flows have public good properties that make it difficult to measure marginal values of use. Volumetric water pricing can also be difficult to implement in practice because of costs associated with measuring water use. Tsur (2009) and Griffin (2001) have published methodological papers outlining the case for marginal cost pricing and practical difficulties with implementation.

Non-volumetric and market-based methods can be used if volumetric water pricing is unfeasible. Johansson (2000) states that area pricing is the most common irrigation water pricing method worldwide. Under an area pricing system, users pay a flat rate for water use per unit of irrigated area. Area pricing methods can encourage efficient water use if charges vary based on crop type, season, irrigation method, and other factors. Market-based methods can address some of the difficulties with identifying efficient volumetric prices by allowing these prices to emerge from market transactions. However, a number of market failures limit the extent to which water markets can identify efficient prices. These include pollution and return flow externalities, high capital investment costs, high transaction costs, and public goods properties associated with some water uses.

The author is only aware of one river basin in Europe, the Guadalquivir River basin in Spain, where volumetric water pricing has been implemented in the agriculture sector (Maestu, 1999). A combination of fixed and volumetric charges is used in this basin. Volumetric prices are tied to energy costs and vary with time of use. Maestu (1999) reports that water use in the irrigation co-operatives that use this price structure is about 2000 m^3/hectare less than in comparable irrigation co-operatives without volumetric water pricing.

3 Methodological framework

A hydro-economic modelling approach is used to estimate the impacts of water pricing policies that meet the water pricing objectives of the WFD. The approach includes a river basin decision support system; methods for predicting water use as a function of water prices; an approach for measuring welfare changes resulting from water price changes; a method for assessing whether environmental flow requirements have been met; an approach for checking groundwater sustainability; an approach for estimating groundwater drawdown at production wells; an optimization approach that is used to identify appropriate water prices; and an uncertainty analysis approach. These components are now described in detail. The case study river basin is also described.

3.1 River basin decision support systems

The approach presented here has been developed to work with a generalized river basin decision support system. In this context, the term "river basin decision support system" refers to a hydrological model developed for the purpose of resolving conflicts over the allocation of water rather than understanding hydrological processes such as floods, runoff impacts due to land use changes, or groundwater flow patterns. These models usually feature simplified representations of hydraulic and hydrological processes: runoff is represented using river gage data, outputs from other models or simplified rainfall-runoff models; river flows are calculated using simple mass balances or flow routing methods; and groundwater is represented using a linear reservoir model or other simplified approach. The models are used to analyze reservoir operations, water rights, and long-term impacts of policy changes, including the impacts of environmental requirements. Some river basin decision support systems incorporate GIS capabilities for managing spatial data. Many include graphical user interfaces to facilitate hands-on modelling exercises with stakeholder participation. Examples of river basin decision support systems include RIBASIM (WL | Delft Hydraulics, 2004), WEAP (SEI, 2005), MIKE BASIN (DHI, 2009), AQUATOOL (Andreu, et al., 1996), and MODSIM (Labadie, 1995).

The river basin decision support system MIKE BASIN is used in this application (DHI, 2009). MIKE BASIN is a commercial product of the Danish hydrological consultancy DHI. The program is implemented as a ArcGIS add-in, and allows the user to develop a node-link modelling network through the ArcGIS interface. Although MIKE BASIN includes some rainfall-runoff, water quality, and hydraulic capabilities, those are not used in this application. The model is run on a daily timestep for a 20-year period using 20 years of historical hydrology (1981-2000) intended to capture a reasonable range of hydrological conditions that may be expected in the future. The hydrological inputs to the model are river inflows at a daily time step that are outputs from the hydrological simulation model MIKE-SHE. Groundwater is represented using a double-layer linear reservoir model, with timeseries recharge inputs from MIKE-SHE. Outflows from groundwater to surface water take place when groundwater levels are above threshold values (ENM Ltd., 2008).

Reservoir storage is limited in the case study basin and therefore few options are available for altering river flows apart from changes to water abstraction patterns. Surface water is allocated on an upstream-downstream basis; upstream demands are satisfied before downstream demands, and upstream demands are not reduced in periods of shortage to reduce impacts on downstream users. Apart from a small sub-basin that is regulated by a reservoir, the basin is operated as an unregulated system where water price is the only tool available to regulate demand. If supplies are not sufficient to meet demands, shortages are recorded.

The runoff and groundwater recharge timeseries values prepared using the MIKE SHE hydrological model package are driven by historical meteorological data and calibrated using river discharge measurements. The meteorological data include daily and/or monthly measurements of rainfall from 21 stations and mean monthly temperatures from 11 stations. The Hargreaves method is used for the estimation of reference evapotranspiration. Meteorological data are distributed spatially using Thiessen polygons. The model is calibrated using mean monthly flows of 3 hydrological years (1995-1998) at the Krinides gauging station, which measures the runoff of more than 80% of the Aggitis catchment area. The rainfall-runoff representation includes a snowmelt module (ENM Ltd., 2005).

The MIKE BASIN irrigation module is used to simulate agricultural water use. The MIKE BASIN irrigation module is based on the FAO-56 irrigation water use methodology (FAO, 1998). Crop water requirements are calculated based on meteorological data and user-specified crop information such as crop coefficients and growth stage lengths. Soil water storage and uptake of water from soil are also modeled based on the FAO-56 approach. Crop yields as a function of water use are modeled using the FAO-33 approach (FAO, 1979), which links yields to cumulative water supply over the growing season.

3.2 Case study basin

The Aggitis River basin in northern Greece is used as the case study basin. The Aggitis basin is part of River Basin District GR11, one of 14 water district areas delineated by the Greek government for the purpose of complying with the requirements of the WFD. The Aggitis River is a tributary of the Strymonas River, which is the principal river around which GR11 is organized. The total area of the study basin is about 2300 km^2. The climate is Mediterranean, with cool, wet winters and hot, dry summers.

The river basin model of the Aggitis is adapted from a MIKE BASIN representation of GR11 developed by DHI and local project partners in Greece as part of an effort to comply with WFD requirements (ENM Ltd., 2008). The basin is divided into 16 subcatchment areas, each of which is a source of surface runoff as well as an independent groundwater aquifer. 59 anthropogenic water users are included in the representation, including 19 urban/domestic water use locations, 15 irrigation locations, 12 livestock water use sites, 11 industrial water use sites, and 2 tourism locations. A small reservoir regulates seasonal flows in the upper basin. The reservoir is operated for flood control and to meet demands of two downstream irrigation locations, both of which take water directly from the reservoir; instream releases only take place in order to maintain the reservoir's flood pool. There are no hydropower facilities in the catchment. A schematic of the basin is shown in Figure 3.1.

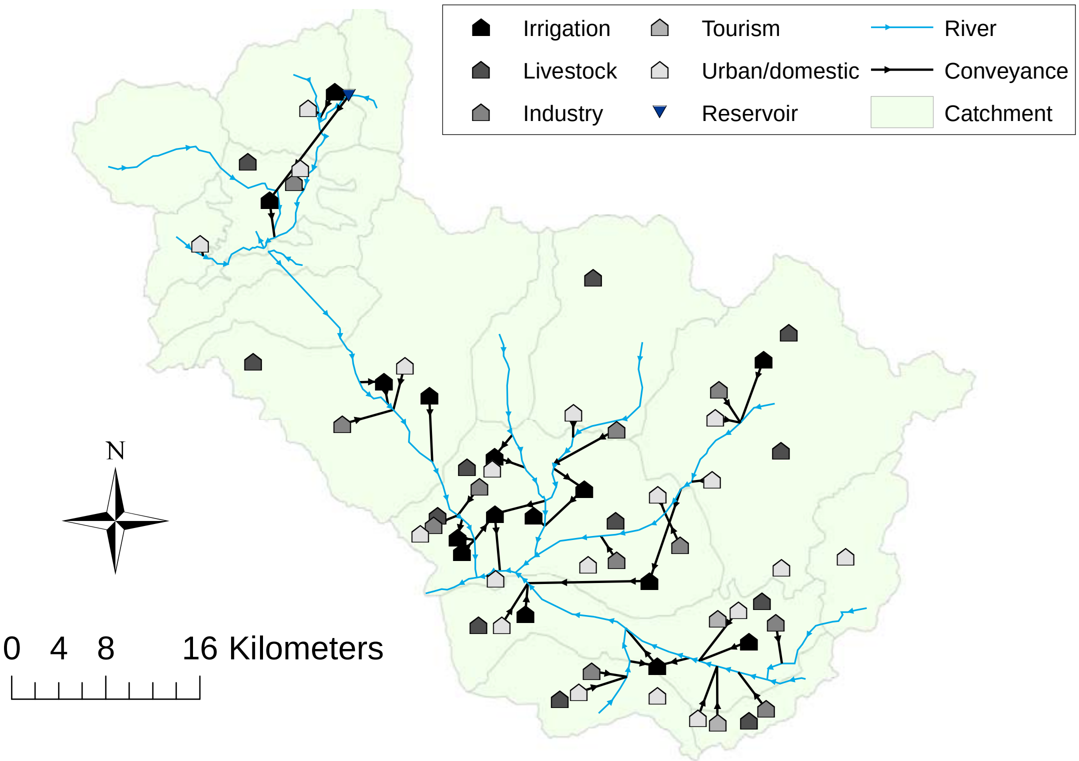

Figure 3.1: Aggitis River basin MIKE BASIN model schematic

This study uses a data set associated with the case study river basin. The data set was assembled during 2007 and is representative of conditions in that year. The case study data set includes information about irrigation, urban/domestic, industry, livestock, and tourism water uses.

3.3 Environmental flow requirements

The central purpose of the WFD is the protection of water resources within the EU, with the goal of achieving good surface water and groundwater status. The WFD defines surface water status in terms of ecological and chemical indicators. These include metrics related to aquatic flora, benthic invertebrates, fish, hydrological regime, river continuity, morphological conditions, thermal conditions, oxygenation levels, salinity, acidification status, nutrient conditions, and pollutant concentrations. This analysis only considers the cost of meeting WFD requirements with respect to hydrological regime. In the WFD lexicon, hydrological regime is considered an indicator of ecological status.

The term “hydrological regime” refers to the pattern of a river’s flow quantity, timing, and variability (Poff et al., 1997). An analysis limited to hydrologic regime may have considerable value because other parameters used to assess ecological status are impacted by the hydrological regime; indeed, it has been argued that the hydrological regime is a “master variable” affecting the distribution and abundance of species in the river ecosystem (e.g., Power et al., 1995).

The WFD status definitions provide considerable latitude for developing flow standards or other instruments in order to achieve a hydrological regime with “good” ecological status. The determination of environmental flow standards is a subject of scientific debate and generally requires detailed analysis of local conditions (Smakhtin et al., 2004; Arthington et al., 2006). In the absence of detailed stream survey information at a basin-wide scale, the method proposed by Arthington et al. (2006) can be used to make a preliminary estimate of the ecological status of the hydrological regime. This analysis uses Arthington’s method for a basin-scale estimate with the understanding that a more detailed assessment of local hydrological conditions is needed.

Arthington recommends a four-step process for developing environmental flow requirements:

1. Select ecologically meaningful flow variables that capture natural flow variability.
2. Develop frequency distributions for each flow variable in each stream reach of interest.
3. Compare frequency distributions from flow-modified conditions to unmodified conditions.
4. Develop flow-response relationships for each flow variable that indicate how the variable is related to indicators of ecological conditions.

Ecologically meaningful flow variables can include measures of magnitude; the frequency, timing, and duration of flow events; rates of change from one flow condition to another; and the temporal sequencing of flow conditions. Poff et al. (1997) advise that “flow history” plots of unmodified stream flows be developed to determine which flow variables may be of interest. Figure 3.2 shows flow history plots of unmodified flows for two reaches in the Aggitis basin. The unmodified flow hydrographs were developed by running the MIKE BASIN model of the basin with all anthropogenic water demands set to zero. The MIKE BASIN model was run on a daily time step for 20 years (01-Oct-1980 to 30-Sep-

2000). The axis labeled "Day of Water Year" refers to an October-September hydrological year.

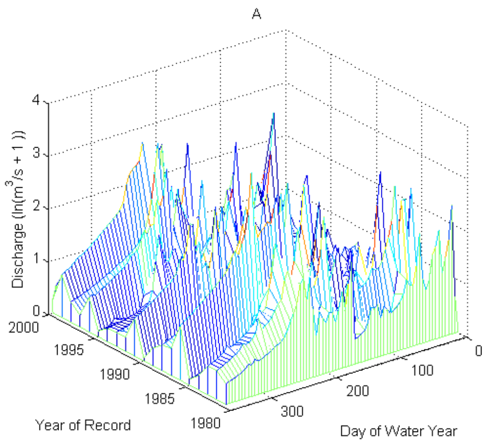

Figure 3.2: Flow history plots for two reaches in case study basin

The flow history plots suggest that the same flow variables could be important for each catchment. The following three flow characteristics appear to be important:

1. The flow pattern is seasonal with a peak in the spring, probably corresponding to the spring snowmelt season. The hydrograph appears to climb suddenly in the early spring and then trail off gradually over the rest of the spring and summer.
2. The flow pattern exhibits annual variation. Although the seasonal pattern of spring peak flows is consistent from year to year, there are considerable differences in the magnitudes of flows from year to year.
3. The flow pattern is flashy and indicates that flow volumes rise and fall rapidly in response to storm events.

This analysis only considers the first of the above flow characteristics. Because there are no large reservoirs in the study area capable of multi-year storage, annual patterns are not disrupted significantly by anthropogenic water use. The lack of significant reservoir storage also means that the impact of storm events is not diminished by human water use. However, abstraction for anthropogenic use can disrupt seasonal flow patterns, particularly during the irrigation season. Therefore, a flow variable (or variables) should be identified to measure the extent to which modified seasonal flows match unmodified flow patterns.

To quantify seasonal flow patterns, monthly flow magnitudes are measured and compared to unmodified conditions. The seasonal pattern of spring runoff, with the sharp increase in early spring followed by a slow decline over the rest of the spring and summer, is captured by the sequence of monthly flow volumes. If the distribution of monthly flow volumes in modified conditions matches the distribution in unmodified conditions, then the seasonal pattern of flows should also be similar to the unmodified seasonal pattern. If a 20-year sequence of monthly flow volumes is sorted by volume in ascending order, the resulting sequence is interpreted as a cumulative distribution function (CDF) of flow probabilities, as outlined in Equation 3.1.

$$\mathbf{CDF}_{jm} = \left[q_{p0.05}\ q_{p0.10}\ \cdots q_{p1.00} \right]' \ = \text{20 x 1 vector of flow volumes for reach j and month m}$$

(sorted in ascending order)

where

$j =$ river reach

$m =$ month (3.1)

$q_{p0.05} =$ monthly flow volume that is not exceeded in 5% of years

$q_{p0.10} =$ monthly flow volume that is not exceeded in 10% of years

$q_{p1.00} =$ monthly flow volume that is never exceeded

Figure 3.3 compares CDFs of modified and unmodified flows for the same reaches that were presented in Figure 3.2. Both plots indicate that the distribution of modified flow volumes differs significantly from the unmodified distribution.

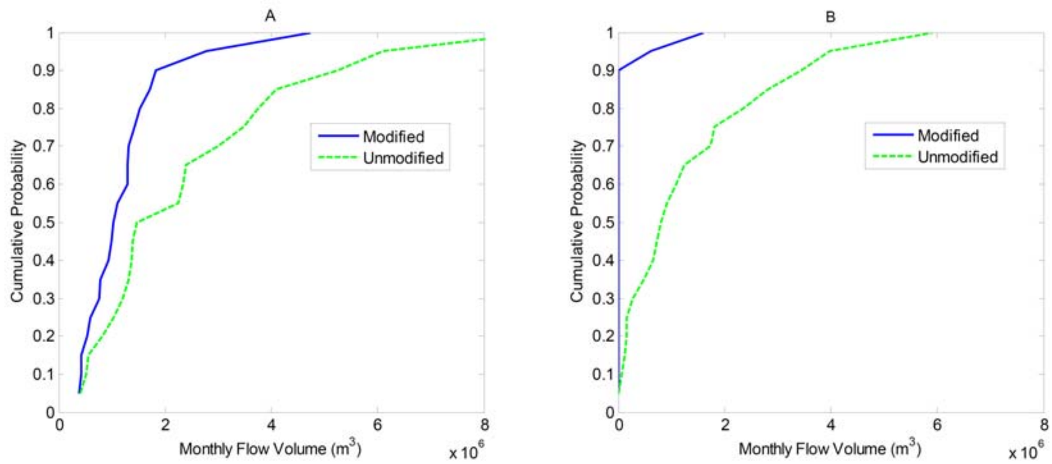

Figure 3.3: Comparison of modified and unmodified CDFs.

The final step in the Arthington methodology is to develop flow-response relationships for each flow variable that indicate how the variable is related to indicators of ecological conditions. A flow-response relationship can be thought of as a performance function (e.g., Loucks, 2006) because it relates the performance of one aspect of the ecosystem (the hydrologic regime) to a measurable variable. In this case, the objective is to link the status of the hydrological regime to the distribution of monthly flow volumes. The maximum vertical difference between CDFs for modified and unmodified conditions, as outlined in Equation 3.2, is used to estimate the state of the hydrological regime.

$$\begin{aligned}
&\text{Let } \mathbf{CDF}_{\text{unmod}} = \text{ 20x1 vector of unmodified monthly flow volumes} \\
&\text{(sorted in ascending order)} \\
&\text{Let } \mathbf{p}_{\text{unmod}} = \text{ 20x1 vector of cumulative probabilities corresponding to entries of } \mathbf{CDF}_{\text{unmod}} \\
&\text{Then } \mathbf{p}_{\text{unmod}} = \left[0.05\ 0.10\ \ldots\ 1.00\right]' \\
&\text{Let } \mathbf{p}_{\text{mod}} = \text{ 20x1 vector of cumulative probabilities for modified flow distribution} \\
&\text{corresponding to entries of } \mathbf{CDF}_{\text{unmod}} \text{ (i.e., } \mathbf{p}_{\text{mod}}(1) = \text{ probability that modified flow volume} \\
&\text{will be less than or equal to } \mathbf{CDF}_{\text{unmod}}(1)) \\
&\text{Let } \sigma = \text{ ecological status} \\
&\text{Then } \sigma = \max\left[abs\left(\mathbf{p}_{\text{unmod}} - \mathbf{p}_{\text{mod}}\right)\right] \\
&\text{where } abs = \text{ absolute value}
\end{aligned} \qquad (3.2)$$

For Reach A in Figure 3.3, the maximum difference between the August CDFs for the modified and unmodified states is about 0.37, and occurs in the vicinity of $2{\cdot}10^6$ m^3. For Reach B, the maximum difference between the August CDFs for the modified and unmodified states is about 0.85, and occurs in the vicinity of 0 m^3.

The above approach is used to assess the ecological status of water allocation plans identified in MIKE BASIN model runs using the following steps.

1. At the conclusion of each model run, cumulative distributions of monthly flows are computed for all reaches and months.
2. Cumulative distributions are compared to cumulative distributions of unmodified monthly flows for all reaches and months.
3. Ecological status levels for all reaches are estimated using Equation 3.2.
4. The ecological status level of the water allocation plan is set equal to the status level representing the tenth percentile of reaches in the study area.

In other words, the reach with a status level equal to or worse than 90% of the other reaches sets the status for the entire study area.

3.4 Groundwater sustainability

The WFD requires that long-term rates of groundwater extraction not exceed rates of recharge. Therefore, a method is necessary to check whether groundwater use predicted in this study is sustainable.

Long-term impacts of groundwater pumping are evaluated using the hydrological model of the case study basin. The hydrological model of the case study basin in divided into 16 subcatchment areas. Each subcatchment area is assumed to function as an independent groundwater body that is represented using a double-layer linear reservoir model. A schematic of the double-layer linear reservoir model is shown in Figure 3.4. In the model of the case study basin, the stream seepage and spill pathways are not active.

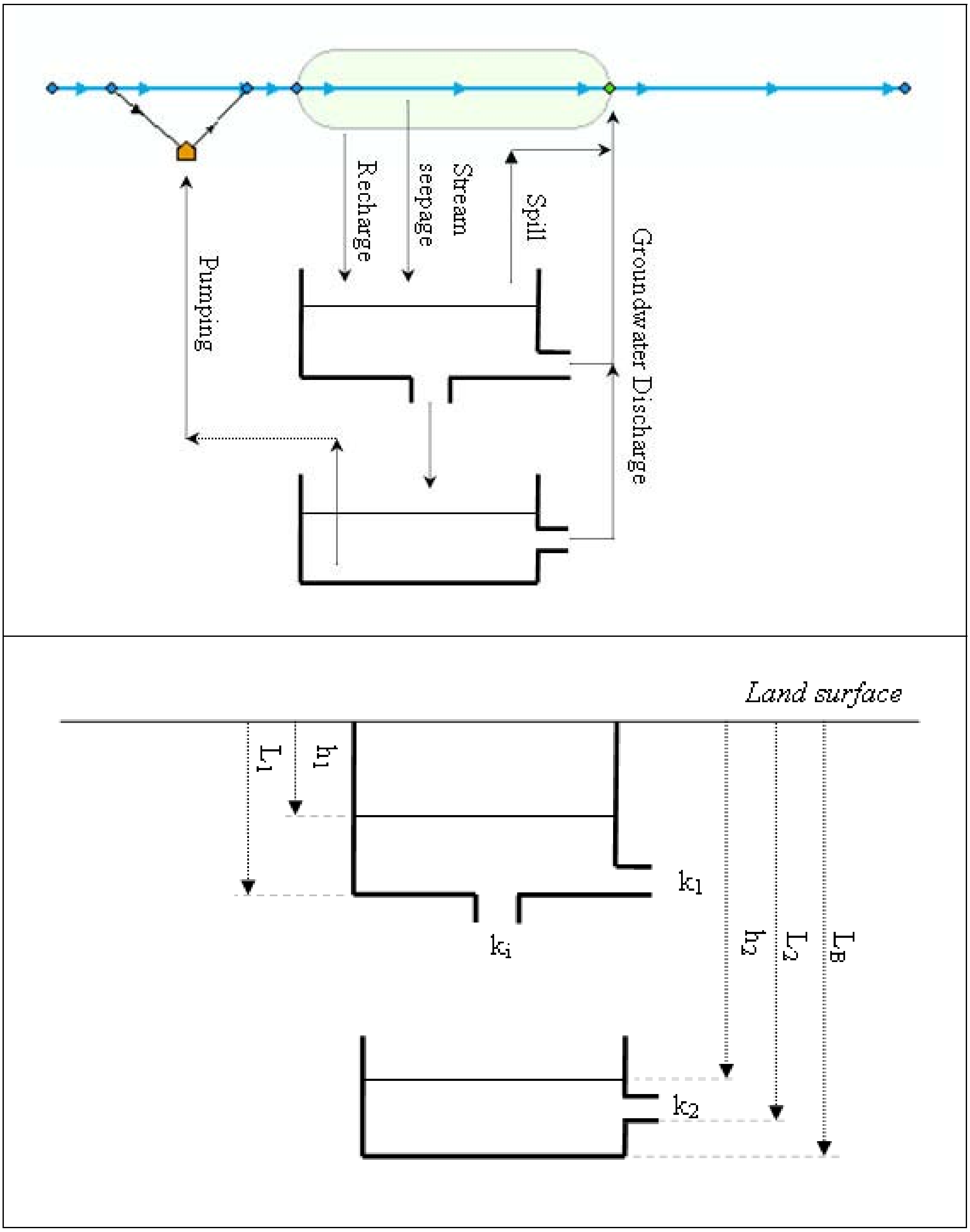

Figure 3.4: Linear reservoir schematic (from MIKE BASIN manual)

In the double-layer representation, groundwater pumping is assumed to extract water from the deep layer but not the shallow layer. The deep layer is modeled as a prismatic tank, with the surface area of the tank equal to the subcatchment surface area. There is no simulation of flow through porous media. The mass balance of the deep layer is as follows:

$$
\begin{aligned}
&\text{if } h_1 < L_1 \\
&\quad I = A \cdot k_i \cdot \left(L_1 - h_1\right) \\
&\text{else} \\
&\quad I = 0 \\
&\text{if } h_2 < L_2 \\
&\quad O = A \cdot k_2 \cdot \left(L_2 - h_2\right) + Q_{pumping} \\
&\text{else} \\
&\quad O = Q_{pumping} \\
&\text{where} \\
&h_1 = \text{ depth to surface of shallow layer (m)} \\
&L_1 = \text{ depth to shallow layer outlet (m)} \\
&I = \text{ inflow to deep layer (m}^3\text{/s)} \\
&A = \text{ surface area of subcatchment (m}^2\text{)} \\
&k_i = \text{ time constant controlling rate of discharge from shallow layer to deep layer (d}^{-1}\text{)} \\
&O = \text{ outflow from deep layer (m}^3\text{/s)} \\
&h_2 = \text{ depth to surface of deep layer (m)} \\
&L_2 = \text{ depth to deep layer outlet (m)} \\
&k_2 = \text{ time constant controlling rate of discharge from deep layer to river (d}^{-1}\text{)} \\
&Q_{pumping} = \text{ groundwater pumping rate (m}^3\text{/d)}
\end{aligned}
\tag{3.3}
$$

A third time constant controls discharge from the shallow layer to the river. Time constant and outlet depth estimates are based on simulations of groundwater flow developed using a MIKE SHE model of the Aggitis basin (ENM, Ltd., 2005).

The depth to the deep layer is used in this study as an indicator of whether groundwater pumping is occurring at a sustainable rate. After water demands have been identified using the methods described in this section, the hydrological model is run with these demands active for the 20-year simulation period. At the conclusion of the simulation period, the final value of the depth to deep groundwater is retrieved. A groundwater sustainability status parameter is then estimated as follows:

$$\sigma_2 = \max\begin{pmatrix} final\_depth_1 - outlet\_depth_1, ..., final\_depth_k - outlet\_depth_k, ..., \\ final\_depth_M - outlet\_depth_M \end{pmatrix}$$

where

$k =$ subcatchment index

$M =$ number of subcatchments (3.4)

$\sigma_2 =$ groundwater sustainability status

$final\_depth_k =$ final depth to deep layer for catchment k (m)

$outlet\_depth_k =$ depth to deep layer outlet for catchment k (m)

If the value of the groundwater sustainability status parameter is greater than 0.1 m, then the rate of groundwater pumping is considered unsustainable. The value of 0.1 m is used in order to allow for a small amount of temporary groundwater mining as a drought mitigation strategy.

3.5 Estimating groundwater drawdown

In the final investigation presented as part of this study, an energy price is used to control groundwater use by controlling the cost of groundwater pumping. Because the energy required to pump groundwater is a function of both flow and head, a method is needed for estimating the impact of groundwater pumping on drawdown.

To estimate how the depth to groundwater changes as a function of groundwater abstraction, we use the Cooper-Jacob equation (Fetter, 2001). The Cooper-Jacob equation is valid for pumping from an infinitely extended, homogeneous and confined aquifer with no recharge. If detailed information on the hydrogeology and wellfields in the study region was available, more precise estimation approaches could be used.

$$h_{\circ} - h = \frac{Q}{4 \cdot \pi \cdot T} \cdot \ln\left(\frac{2.25 \cdot T \cdot t}{r^2 \cdot S}\right)$$

where

$h_{\circ} =$ initial elevation of water table relative to land surface (m)

$h =$ final elevation of water table relative to land surface (m)

$Q =$ groundwater pumping rate (m^3/d) (3.5)

$T =$ transmissivity (m^2/d)

$r =$ well radius (m)

$S =$ storativity

$t =$ time since pumping began (d)

The Cooper-Jacob equation requires estimates of initial groundwater table elevations, pumping rates, transmissivities, well radii, storativities, and pumping times. Initial groundwater table elevations, pumping rates, transmissivities, well radii, storativities, and pumping times are assumed to be constant throughout the case study area. The initial groundwater table elevation is assumed to equal 5 m below terrain at all water use locations (ENM, Ltd., 2005). Transmissivity is assumed to equal 216 m^2/day (Panilas, 1998). The radius of each well is assumed to equal 0.1 m. Storativity is assumed to equal 1E-4.

The Cooper-Jacob equation requires estimates of flow rates at individual wells in order to estimate drawdown. The baseline data set includes information about annual and monthly water use volumes but no information about flow rates. In addition, the methods that are used to predict how water users will respond to water price changes predict volumes of water that will be used, not flow rates. Average flow rates are estimated as follows:

$$Q = \frac{V}{n \cdot L_p}$$

where

$V =$ annual volume pumped (m^3) (3.6)

$L_p =$ average number of days of pumping per year (d)

$n =$ number of wells

The Cooper-Jacob equation estimates drawdown at an instantaneous point in time. To estimate average drawdown over a given time period, the Cooper-Jacob equation can be integrated with respect to time.

$$\text{average drawdown (m)} = \frac{1}{L_d} \cdot \frac{Q}{4 \cdot \pi \cdot T} \cdot \int_0^{L_d} \ln\left(\frac{2.25 \cdot T \cdot t}{r^2 \cdot S}\right) dt$$

$$= \frac{1}{L_d} \cdot \frac{Q}{4 \cdot \pi \cdot T} \cdot \left(t \cdot \ln\left(\frac{2.25 \cdot T \cdot t}{r^2 \cdot S}\right) - t\right)_0^{L_d} \tag{3.7}$$

where

$L_d =$ length of drawdown period (d)

If a small positive value is substituted for zero as the lower limit of integration, the expression for average drawdown becomes:

$$ad(Q, L_d) = \frac{Q}{4 \cdot \pi \cdot T} \cdot \left(\ln\left(\frac{2.25 \cdot T \cdot L_d}{r^2 \cdot S}\right) - 1\right) \tag{3.8}$$

where $ad(Q, L_d) =$ average drawdown (m)

No information is available about the number of wells at each water use location. According to local project partners in the case study basin, 75 m is a typical drawdown value. Therefore, it is assumed that average drawdown does not exceed 75 m at any well in the baseline data set. The number of wells at each location is estimated by identifying the number of wells required to limit drawdown to 75 m when water is pumped at the rate observed in the baseline data set.

$$\text{n} = \left\lceil \frac{\dfrac{V}{L_p \cdot ad\_max}}{4 \cdot \pi \cdot T} \cdot \left(\ln\left(\frac{2.25 \cdot T \cdot L_d}{r^2 \cdot S}\right) - 1\right) \right\rceil$$

where (3.9)

$ad\_max = 75$ m

The expression for average drawdown is used to estimate the cost of pumping a given volume of water over a single growing season. This estimate assumes that drawdown over the growing season can be approximated by average drawdown.

$$\text{pumping cost (€)} = pe_{policy} \cdot \frac{\rho \cdot g}{\eta} \cdot V \cdot \left(-h_{\circ} + ad(Q, L_d)\right)$$

where

$pe_{policy} =$ energy policy price (€/kWh)

$\rho =$ density of water (kg/m^3) (3.10)

$g =$ acceleration due to gravity (m/s^2)

$\eta =$ pumping efficiency

$V =$ volume of water pumped (m^3)

This expression is a quadratic function of water use. A pumping efficiency of 0.7 is assumed at all locations.

Many assumptions have been made in order to develop this approach for estimating groundwater pumping costs. However, the important point to be made is that a reasonable quadratic cost function has been developed that can be used to estimate the cost of water supply as a function of water use. Any function developed to estimate groundwater pumping costs will have a quadratic form regardless of the assumptions used to develop it.

3.6 Predicting water use as a function of water prices

In this study, urban/domestic water users are modeled as utility-maximizing consumers, while irrigation, industry, livestock, and tourism water users are modeled as profit-maximizing producers. Methods used to predict how urban/domestic, irrigation, and industry water users will respond to water prices are presented in this section. Methods for estimating welfare changes as a result of water price changes are also presented.

Because of the importance of the agriculture sector (about 90% of water use in the case study basin is by agriculture), two different approaches are used to model how irrigation water users will respond to water price changes. One approach is based on the residual imputation method (Young, 2005; Griffin, 2006). The other is based on Positive Mathematical Programming (Howitt, 1995a; Howitt, 1995b).

Methods used to model livestock and tourism water users are not presented here because total water use for these use types is much less than for other types. The approach used to model livestock and tourism water users is similar to the approach used for industry.

3.6.1 Irrigation water use: Residual imputation method

The residual imputation method estimates marginal values of water use by assuming that all input costs other than water costs are known, and that the marginal costs of these inputs are equal to marginal values of these inputs in production (i.e., that the selection of input levels is consistent with the principle of profit maximization). The marginal value of water is then assumed to equal to the difference between the gross production value and all costs of production apart from water costs. In other words, the method uses the *residual* of the gross production value (after all other costs have been subtracted) to *impute* the value of water. A unit marginal value of water can then be estimated by dividing the residual value of water by the total amount of water use.

$$p_w^* = \frac{p_y \cdot f(w, \mathbf{x}) - \mathbf{p}_x \cdot \mathbf{x}}{w} \tag{3.11}$$

where

$p_y =$ producer price

$p_w^* =$ unit marginal value of water

$w =$ water input

$\mathbf{p}_x =$ vector of non-water input prices

$\mathbf{x} =$ vector of non-water inputs

The baseline data set that is available for the case study basin includes information about producer prices, maximum yields, subsidies, and input costs. Input cost data include fertilizer, pesticide, seed, fuel, and mechanical collection costs. Labor costs are assumed to be part of mechanical collection costs. Irrigation fees, which are charges on an area basis, are also included in the baseline data set. All of these data are assumed to be constant throughout the study area (i.e., none vary from one irrigation water use location to the next). Input costs are priced on an area basis.

The baseline data set does not include estimates of land rents. In the case study area, it is common for land to be rented with an agreement for profits to be split between landowner and renter, with 1/3 of profits going to the landowner (Kitsopanidis et al., 2003). This assumuption was used to estimate land rent costs in this study. The procedure used to estimate land rental costs is outlined below.

$$c\_land_{ij} = \frac{py_i \cdot \overline{y}_{ij} + s_i - \text{sum}(\mathbf{c}_i)}{3} \tag{3.12}$$

where

$i =$ crop index

$j =$ location index

$c\_land_{ij} =$ estimated land rental cost (€/ha)

$py_i =$ producer price for crop i in baseline data set (€/tonne)

$\overline{y}_{ij} =$ average yield for crop i at location j (tonnes/ha)

$s_i =$ subsidy for crop i in baseline data set (€/ha)

$\mathbf{c}_i =$ vector of input costs for crop i in baseline data set (€/ha)

The procedure outlined above uses a quantity called average yield to estimate the marginal value of water. Actual yields from the base year are not availabe as part of the baseline data set, which only includes information about maximum yields. It seems unlikely that maximum yields were observed for all crops at all locations during the base year. To develop a more realistic estimate of yields for use with this approach, crop yields are estimated using the irrigation model that is part of the MIKE BASIN model package. The MIKE BASIN irrigation model estimates crop yields using an approach that is based on FAO-33 (FAO, 1979). In this approach, yield is assumed to be a function of the ratio of actual crop evapotranspiration to potential crop evapotranspiration. Average yield is estimated by running MIKE BASIN and the MIKE BASIN irrigation module using the 1981-2000 data set and taking the average of yields estimated during the simulation period. The MIKE BASIN irrigation model estimates crop yield using the following formula:

$$Y_a = Y_{\max} \cdot \left(1 - k_y \cdot \left(1 - \frac{AET}{PET}\right)\right)$$

where

$Y_a =$ estimated actual yield

$Y_{\max} =$ maximum yield from baseline data set (3.13)

$k_y =$ crop yield coefficient

$AET =$ actual crop evapotranspiration over entire growing season

$PET =$ potential crop evapotranspiration over entire growing season

Crop yields estimated using the MIKE BASIN irrigation model are less than maximum yields in some years because of water availability constraints that limit irrigation water use. The use of average crop yields estimated using the MIKE BASIN irrigation model is not consistent with the 2007 baseline data set because these yields represent an average of conditions simulated during the period from 1981 to 2000 and not conditions that were observed in 2007. However, yield data from the 2007 data set consist of maximum yields, not observed yields. It is not clear that maximum yields were observed for all crops and at all locations in 2007. Average yields estimated using the MIKE BASIN irrigation model may be a reasonable approximation of yields observed in 2007.

Estimates of water use by crop type are also not available as part of the baseline data set. The baseline data set includes estimates of water use by water use location but not by crop type. Estimates of water use by crop type are also developed using the MIKE BASIN package.

To estimate average annual water use for each crop and at each irrigation location, the MIKE BASIN hydrological model of the case study river basin is run for a 20-year period using historical hydrological and meteorological input data from the period 1981-2000. During this model run, the MIKE BASIN irrigation model is active and irrigation water use is computed based on the soil water balance and estimates of crop evapotranspiration. The method used to compute crop water use is based on FAO-56 (FAO, 1998). The model runs on a daily time step, with the soil moisture balance and crop water use computed every day. During some days, irrigation water use does not equal demand because of water availability constraints. Average annual water use is estimated by taking the average of the total amount of water use simulated during each year.

The method used to estimate annual water use by each crop is not consistent with the baseline data set because the baseline data set is based on values observed in 2007 while water use is based on an average of values estimated between 1981 and 2000. However, it is possible to compare total estimated use at each location to total use at each location observed in the baseline data set. These totals are compared in Figure 3.5 for all irrigation water use locations. The figure suggests that estimated average water use during the 1981-2000 period is similar to observed water use in 2007 for most water use locations. It was not possible to estimate water use in 2007 using the MIKE BASIN irrigation model because hydrological data were not available for this year. With the exception of water use locations 387 and 389, estimated 1981-2000 water use appears to be reasonably close to observed 2007 water use. Because total water use at locations 387 and 389 is a small fraction of total use for the entire basin, average values from the 1981-2000 simulation period are appropriate for use with the 2007 baseline data set.

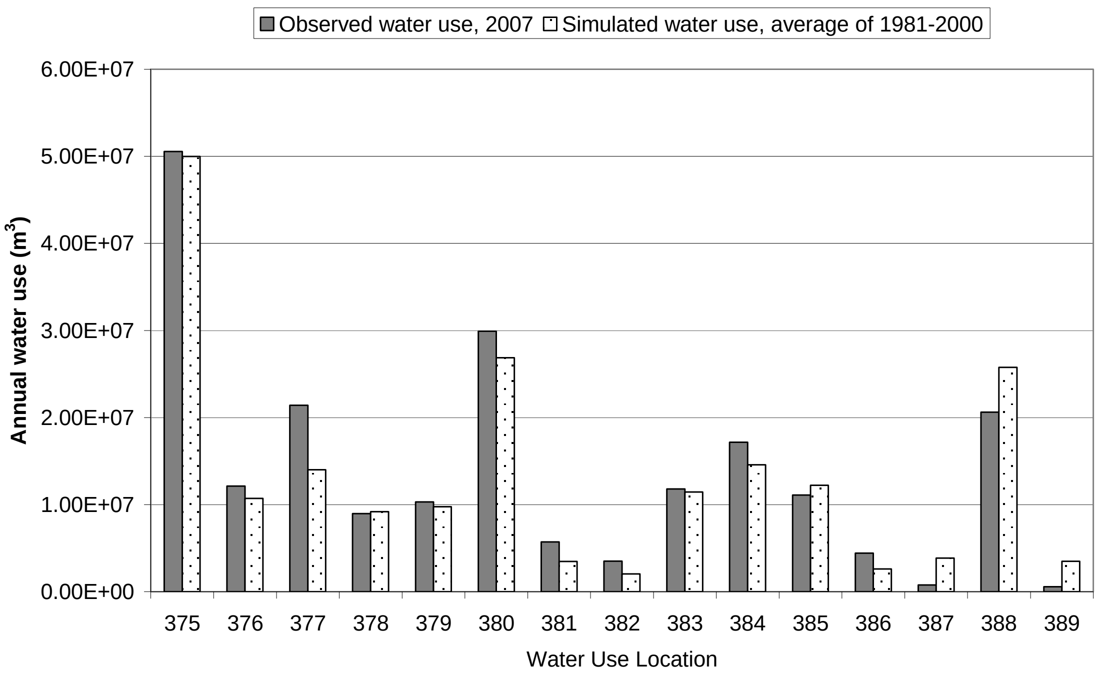

Figure 3.5: Comparison of simulated and observed irrigation water use.

With these assumptions, it is possible to estimate marginal values of water use using the residual impuation method:

$$pw_{ij}^{*} = \frac{py_i \cdot \bar{y}_{ij} + s_i - c\_land_{ij} - \text{sum}(\mathbf{c}_i) + c\_irrigation_i}{w\_irrigation\_base_{ij}}$$

where

$pw_{ij}^{*} =$ marginal value of water identified by residual imputation (€/m^3) (3.14)

$w\_irrigation\_base_{ij} =$ average annual water use from MIKE BASIN simulation (m^3)

$c\_irrigation_i =$ irrigation fee for crop i (€/hectare)

Irrigation fees are not included in the estimate of the marginal value of water because the marginal value of water is assumed to include these fees. This assumption implies that irrigation fees do not capture the full value of water.

The marginal value of water identified using the residual imputation method should be interpreted as an upper bound on the actual marginal value. It is difficult to quantify all non-water input costs, particularly costs of owner inputs such as entrepreneurial creativity and management ability, so it is likely that a portion of the value attributed to water is actually the result of another input factor.

The marginal values of water use identified using the residual imputation method are used to predict how irrigation water users will react to water price changes. If water prices are less than these marginal values, it is assumed that land allocated to these crops in the baseline scenario remains allocated to the same crops. If water prices exceed marginal values, it is assumed that the land allocated to these crops is converted to dryland agriculture. The procedure is summarized below:

for $i = 1:N$

if $p\_policy_w \le pw_{ij}^*$

$A_{ij}^{'} = A_{ij}$

else

$A_{ij}^{'} = 0$

$A\_dryland_j = A\_dryland_j + A_{ij}$ (3.15)

where

$N =$ number of crops

$p\_policy_w =$ simulated water price (policy variable) (€/m^3)

$A_{ij} =$ area of crop i at location j in baseline data set (ha)

$A_{ij}^{'} =$ land allocated to crop i at loction j as a result of policy water price (ha)

$A\_dryland_j =$ area allocated to dryland production at location j (ha)

The procedure outlined above is essentially binary. If the simulated water price is less than or equal to a crop's marginal value of water use identified using residual imputation, then land use for that crop is set to the level observed in the baseline data set. If the simulated water price is greater than a crop's marginal value of water use identified using residual imputation, then land use for that crop is set to zero and the area of the crop observed in the baseline data set is assumed to be converted to dryland production.

When the residual imputation method is used, welfare changes are estimated by measuring changes in net benefits. For each irrigation water use location, net benefits are computed as follows:

$$NB_j = \sum_{i=1}^{N} \left(py_i + ps_{ij} \right) \cdot \overline{y\_sim_{ij}} \cdot A_{ij}^{'} - c\_land_{ij} \cdot A_{ij}^{'} - p\_policy_w \cdot \overline{w\_sim_{ij}^{'}} - sum\left(A_{ij}^{'} \cdot \mathbf{c}_i \right)$$

where

$NB_j =$ net benefit at location j (€) (3.16)

$\overline{y\_sim_{ij}} =$ average yield of crop i at location j during simulation period (tonnes/ha)

$\overline{w\_sim_{ij}} =$ average annual water use by crop i at location j during simulation period (m^3)

Yields and water use levels used in the net benefit computation presented in the above equation are averages of MIKE BASIN results. Average MIKE BASIN results are used because the residual imputation method, as presented here, does not predict water use or yields. The method is used to predict allocations of land

and other non-water inputs. Allocations of land predicted using the method are then written to MIKE BASIN, which is used to estimate resulting yields and water use.

In the baseline data set, subsidy values are given in units of €/hectare. It is not clear whether these subsidies are paid regardless of crop production. For example, it seems unlikely that a subsidy would be paid if land was allocated to production of a crop but no other inputs were allocated (which would mean that nothing would be produced). To operationalize subsidies within the framework being described here, it is assumed that subsidy values vary linearly with production, with the given subsidy values equal to subsidies that would be paid given average yields. This assumption allows for estimation of subsidy value in units of €/tonne produced, as shown below.

$$ps_{ij} = \frac{s_i}{y_{ij}} \tag{3.17}$$

where

$ps_{ij} =$ subsidy benefit (€/tonne)

In addition to net benefits to irrigation water users, it is also necessary to estimate supply costs. Supply costs exist because of fixed capital and operating costs associated with the construction and operation of irrigation projects. Supply costs also exist because of groundwater pumping costs. Because volumetric prices are equal for surface water and groundwater, it is assumed that the basin authority implementing the pricing policy assumes responsibility for pumping costs to ensure that the marginal costs of groundwater and surface water use are the same. Total net benefits at each irrigation water use location are then estimated as follows:

$$TNB_j = NB_j - c\_capital_j - c\_operating_j - c\_pumping \cdot \sum_{i=1}^{N} \overline{gw\_sim_{ij}^{'}}$$

where

$TNB_j =$ total net benefit at location j (€ / year)

$c\_capital_j =$ capital cost at location j (€ / year) (3.18)

$c\_operating_j =$ operating cost at location j (€ / year)

$c\_pumping =$ groundwater pumping cost (€/m^3)

$\overline{gw\_sim_{ij}^{'}} =$ average groundwater use by crop i at location j in simulation (m^3/year)

Groundwater use simulated by the MIKE BASIN model depends on the fraction of demand met by groundwater at irrigation water use location. The cost of pumping groundwater is assumed to be constant throughout the case study basin.

3.6.2 Irrigation water use: Positive Mathematical Programming

The Positive Mathematical Programming (PMP) approach was introduced by Howitt (1995a) as a method for calibrating agricultural production functions when limited data are available. The method can be used to parameterize crop production functions that reproduce existing land allocations under the assumption of profit maximization. It was extended to accommodate multiple inputs using the assumption of a constant elasticity of substitution (Howitt, 1995b). Because the method only requires estimates of prices, yields, and input costs, it is suitable for use with the baseline data set available as part of this study.

The central insight of PMP is that observed land allocations are the result of optimizing behavior by farmers. The approach begins by assuming that crop production is a linear function of land, with observed land allocations as constraints. The resulting shadow prices on these constraints are then used to parameterize production and cost functions. If profit maximization is assumed, these production and cost functions will reproduce observed production and input levels, even without land allocation constraints. The approach as applied in this study is now outlined.

The PMP approach used here begins by assuming that crop production is a linear function of land, where production is equal to the product of average yield and area.

$$y_{ij} = \overline{y_{ij}} \cdot A_{ij}^*$$

where (3.19)

$y_{ij} =$ total yield for crop i at location j (tonnes)

$A_{ij}^* =$ allocation of land to crop i at location j (ha)

The profit obtained from production of any crop is as follows:

$$profit_{ij} = py_i \cdot \overline{y_{ij}} \cdot A_{ij}^* + s_{ij} \cdot A_{ij}^* - c\_land_{ij} \cdot A_{ij}^* - \text{sum}\left(A_{ij}^* \cdot \mathbf{c}_i\right)$$

where (3.20)

$profit_{ij} =$ profit of crop i at location j (€)

At any irrigation water use location in the case study area, up to 15 crops are grown. Although each irrigation water use location may consist of more than one independent farm, each location is treated as a single profit-maximizing entity for the purposes of this analysis. Profits are maximized at each location using linear programming with baseline land allocations as constraints. Total water use is also constrained to baseline totals. A small perturbation factor is applied to the individual crop land constraints so that the total land constraint will bind. The objective function and constraints are given below.

$$\max \sum_{i=1}^{N} A_{ij}^* \cdot \left(py_{ij} \cdot \overline{y_{ij}} \cdot + s_i - c\_land_{ij} - \text{sum}(\mathbf{c}_i) \right)$$

$s.t.$

$A_{ij}^* \leq A_{ij} \cdot (1+\varepsilon)$ for all i crops

$$\sum_{i=1}^{N} A_{ij}^* \leq \sum_{i=1}^{N} A_{ij}$$

$$\sum_{i=1}^{N} w_{ij}^* \leq \sum_{i=1}^{N} w\_irrigation\_base_{ij} \qquad (3.21)$$

where

$\varepsilon = 1E-4$

$w_{ij}^* = \overline{w_{ij}} \cdot A_i^*$

$$\overline{w_{ij}} = \frac{w\_irrigation\_base_{ij}}{A_{ij}}$$

In Equation 3.21, the decision variables are the land areas allocated to each crop.

The optimization outlined in Equation 3.21 gives a number of shadow prices.

$$\begin{aligned}
&\lambda_{ij} = \text{ shadow price on land constraint for crop i at location j} \\
&\lambda_land_j = \text{ shadow price on total land constraint for location j} \\
&\lambda_water_j = \text{ shadow price on total water constraint for location j}
\end{aligned} \tag{3.22}$$

These shadow prices are used to parameterize crop production and cost functions. Crop production is assumed to be a function of three inputs: land, water, and an aggregate input that is equal to the sum of all other observed inputs. The aggregate input that is the sum of all other inputs will be referred to as “capital” from this point forward. Capital is measured in currency units (€). The crop production function is assumed to have the following form:

$$\begin{aligned}
&y_{ij} = \alpha_{ij} \cdot (\beta_{1ij} \cdot x_{1ij}{}^{\gamma} + \beta_{2ij} \cdot x_{2ij}{}^{\gamma} + \beta_{3ij} \cdot x_{3ij}{}^{\gamma})^{1/\gamma} \\
&\text{where} \\
&\alpha_{ij} = \text{ scale parameter} \\
&\beta_{1ij}, \beta_{2ij}, \beta_{3ij} = \text{ share parameters} \\
&x_{1ij} = \text{ land input to production of crop i at location j (ha)} \\
&x_{2ij} = \text{ water input to production of crop i at location j (m}^3\text{)} \\
&x_{3ij} = \text{ capital input to production of crop i at location j (€)} \\
&\gamma = \frac{\sigma - 1}{\sigma} \\
&\sigma = \text{ elasticity of substitution}
\end{aligned} \tag{3.23}$$

The above equation is an example of a constant elasticity of substitution (CES) production function. The CES production function was introduced by Arrow et al. (1961) to provide an alternative to Cobb-Douglas and Leontief (fixed factors) production forms. The development of the CES production function was motivated by an international comparison of capital and labor in production. The international comparison suggested that capital and labor, while substitutable, do not always appear to exhibit substitutability patterns that can be described by an elasticity of substitution equal to unity, as implied by the Cobb-Douglas form. Elasticity of substitution is defined as the percentage change in the technical rate of substitution between two inputs that can be expected as a result of a percentage change in the ratio of these inputs.

$$\sigma = \frac{d\left(TRS\right)}{d\left(\frac{x_1}{x_2}\right)} \cdot \frac{\frac{x_1}{x_2}}{TRS} \tag{3.24}$$

where

$TRS =$ technical rate of substitution

$x_1, x_2 =$ inputs

The technical rate of substitution is defined as the rate at which one input can be substituted for another while holding output constant (Perman et al., 1996). For a two-input production process, the technical rate of substitution is derived as follows:

$$\begin{aligned}
&\text{Let } y = f\,(x_1, x_2) \\
&\rightarrow dy = \frac{\partial f}{\partial x_1} \cdot dx_1 + \frac{\partial f}{\partial x_2} \cdot dx_2 \\
&\text{If } dy = 0, \\
&\frac{\partial f}{\partial x_1} \cdot dx_1 = -\frac{\partial f}{\partial x_2} \cdot dx_2 \\
&\rightarrow \frac{dx_1}{dx_2} = -\frac{\frac{\partial f}{\partial x_2}}{\frac{\partial f}{\partial x_1}} \\
&\rightarrow TRS = -\frac{\frac{\partial f}{\partial x_2}}{\frac{\partial f}{\partial x_1}}
\end{aligned} \tag{3.25}$$

The parameters of the CES production function presented are estimated by assuming that the marginal value of each input is equal to the observed cost of that input plus the shadow price associated with the constraint on that input (Howitt, 1995b). In the case of the land input, the shadow price associated with the observed land allocation constraint is added to the marginal value of land.

$$\left(py_i + ps_{ij}\right)\cdot\frac{\partial y_{ij}}{\partial x_{1ij}} = \omega_{1ij}$$

$$\left(py_i + ps_{ij}\right)\cdot\frac{\partial y_{ij}}{\partial x_{2ij}} = \omega_{2j}$$

$$\left(py_i + ps_{ij}\right)\cdot\frac{\partial y_{ij}}{\partial x_{3ij}} = \omega_{3j}$$

where

$\frac{\partial y_{ij}}{\partial x_{1ij}},\frac{\partial y_{ij}}{\partial x_{2ij}},\frac{\partial y_{ij}}{\partial x_{3ij}} =$ marginal products of land, water, and capital

$\omega_{1ij} = c_{1ij} + \lambda_{ij} + \lambda_{1j}$

$\omega_{2j} = c_{2j} + \lambda_{2j}$

$\omega_{3j} = c_{3j}$

$c_{1ij}, c_{2j}, c_{3j} =$ observed costs of land, water, and capital

$\lambda_{ij} =$ shadow price associated with observed allocation of land to crop i at location j

$\lambda_{1j} =$ shadow price associated with total land constraint at location j (3.26)

$\lambda_{2j} =$ shadow price associated with total water constraint at location j

After marginal values have been estimated, the parameters of the CES production are estimated as follows:

$$\frac{\partial y_{ij}}{dx_{1ij}} = \alpha_{ij}\cdot\beta_{1ij}\cdot x_{1ij}{}^{\gamma-1}(\beta_{1ij}\cdot x_{1ij}{}^{\gamma} + \beta_{2ij}\cdot x_{2ij}{}^{\gamma} + \beta_{3ij}\cdot x_{3ij}{}^{\gamma})^{1/\gamma - 1}$$

$$\frac{\partial y_{ij}}{dx_{2ij}} = \alpha_{ij}\cdot\beta_{2ij}\cdot x_{2ij}{}^{\gamma-1}(\beta_{1ij}\cdot x_{1ij}{}^{\gamma} + \beta_{2ij}\cdot x_{2ij}{}^{\gamma} + \beta_{3ij}\cdot x_{3ij}{}^{\gamma})^{1/\gamma - 1}$$

$$\gamma - 1 = -\frac{1}{\sigma}, \frac{1}{\gamma} - 1 = \frac{1}{\sigma - 1} \qquad (3.27)$$

$$\rightarrow \beta_{2ij} = \beta_{1ij}\cdot\frac{\omega_{2j}\cdot x_{1ij}{}^{-\frac{1}{\sigma}}}{\omega_{1ij}\cdot x_{2ij}{}^{-\frac{1}{\sigma}}}, \beta_{3ij} = \beta_{1ij}\cdot\frac{\omega_{3j}\cdot x_{1ij}{}^{-\frac{1}{\sigma}}}{\omega_{1ij}\cdot x_{3ij}{}^{-\frac{1}{\sigma}}}$$

The CES production function is assumed to have the property of constant returns to scale, which allows for the estimation of another of the parameters. The CES production function exhibits constant returns to scale if the beta parameters of the production function sum to unity.

$$\beta_{1ij} + \beta_{2ij} + \beta_{3ij} = 1$$

$$\rightarrow \frac{1}{\beta_{1ij}} = 1 + \frac{\omega_{3j} \cdot x_{1ij}^{-\frac{1}{\sigma}}}{\omega_{1ij} \cdot x_{3ij}^{-\frac{1}{\sigma}}} + \frac{\omega_{2j} \cdot x_{1ij}^{-\frac{1}{\sigma}}}{\omega_{1ij} \cdot x_{2ij}^{-\frac{1}{\sigma}}} \tag{3.28}$$

The parameter α can be estimated by assuming that production equals the observed level of production.

$$\alpha_{ij} = \frac{\overline{y_{ij}} \cdot A_{ij}}{(\beta_{1ij} \cdot x_{1ij}^{\gamma} + \beta_{2ij} \cdot x_{2ij}^{\gamma} + \beta_{3ij} \cdot x_{3ij}^{\gamma})^{1/\gamma}} \tag{3.29}$$

The only parameter of the CES production function that can not be estimated from the baseline dataset is the elasticity of substitution. In this study, the elasticity of substitution was set to 0.5. This value is estimated by assuming that the elasticity of substitution can be used as a calibration parameter that is adjusted so that yields predicted by the CES production function match yields observed in the MIKE BASIN simulation model.

The CES production will not reproduce observed production and input levels unless a non-linear land cost function is estimated. If profit maximization is assumed at a single irrigation water use location, production and input levels are adjusted until marginal profits are equal for all crop types. The marginal values of land use are based in part on shadow prices associated with existing land allocation constraints. Because these constraints are different for each crop type, the shadow prices must be incorporated into the land cost function for each crop type or it will not be possible to equate marginal profits across crop types given observed input costs. A quadratic form is assumed for the land cost function:

$$
\begin{aligned}
& f(x_{1ij}) = a_{ij} \cdot x_{1ij} + 0.5 \cdot b_{ij} \cdot x_{1ij}{}^{2} \\
& \rightarrow f'(x_{1ij}) = a_{ij} + b_{ij} \cdot x_{1ij} \\
& \rightarrow \overline{f(x_{1ij})} = a_{ij} + 0.5 \cdot b_{ij} \cdot x_{1ij}
\end{aligned}
\tag{3.30}
$$

where

$f(x_{1ij}) =$ land cost of crop i at location j (€)

$f'(x_{1ij}) =$ marginal land cost of crop i at location j (€/ha)

$\overline{f(x_{1ij})} =$ average land cost of crop i at location j (€/ha)

$a_{ij}, b_{ij} =$ quadratic land cost parameters for crop i at location j

The difference between the average and marginal cost of land at the observed land allocation is assumed to equal the shadow price associated with the observed land allocation.

$$
\lambda_{1ij} = f'(x_{1ij}) - \overline{f(x_{1ij})} \tag{3.31}
$$

This assumption is reasonable because of the way that shadow prices associated with observed land constraints were estimated. These shadow prices exist because the marginal profits of the different crop types are not equal if production and costs are assumed to be linear functions of land. If observed allocations of land are assumed to be profit-maximizing, as is assumed when using PMP, then it is reasonable to assume that the shadow prices on existing land allocation constraints are equal to the difference between average (linear) and marginal (non-linear) profits.

It is also assumed that land costs associated with the baseline data set are equal to average land costs estimated using a quadratic land cost function.

$$
\overline{f(x_{1ij})} = c\_land_{ij} \tag{3.32}
$$

With this assumption, the parameters of the quadratic land cost function can be estimated.

$$
\begin{aligned}
&\lambda_{1ij} = a_{ij} + b_{ij} \cdot x_{1ij} - a_{ij} + 0.5 \cdot b_{ij} \cdot x_{1ij} \\
&b_{ij} = \frac{2 \cdot \lambda_{1ij}}{x_{1ij}} \\
&c\_land_{ij} = a_{ij} + 0.5 \cdot b_{ij} \cdot x_{1ij} \\
&a_{ij} = c\_land_{ij} - \lambda_{1ij}
\end{aligned}
\tag{3.33}
$$

The CES production function and associated cost functions will reproduce observed production and input levels if profit maximization is assumed and total land and water use are constrained to observed levels. The objective function and constraints are:

$$
\begin{aligned}
&\max \sum_{i=1}^{N} \begin{pmatrix} \left(py_i + ps_{ij}\right) \cdot \alpha_{ij} \cdot (\beta_{1ij} \cdot x_{1ij}{}^{\gamma} + \beta_{2ij} \cdot x_{2ij}{}^{\gamma} + \beta_{3ij} \cdot x_{3ij}{}^{\gamma})^{1/\gamma} - a_{ij} \cdot x_{1ij} - 0.5 \cdot b_{ij} \cdot x_{1ij}{}^{2} \\ -c_{2ij} \cdot x_{2i} - c_{3j} \cdot x_{3ij} \end{pmatrix} \\
&\text{subject to} \\
&\sum_{i=1}^{N} x_{1ij} \leq \sum_{i=1}^{N} A_{ij} \\
&\sum_{i=1}^{N} x_{ij} \leq \sum_{i=1}^{N} w\_irrigation\_base_{ij}
\end{aligned}
\tag{3.34}
$$

The decision variables in the objective function are the input levels x_1, x_2, and x_3.

The CES production function requires estimates of water costs that are consistent with the baseline data set. In the case of water costs, the approach requires an estimate of a volumetric water price. A volumetric price is estimated by dividing the observed water price, which is applied on a land area basis, by the observed unit water use.

$$
c_{2ij} = \frac{c\_irrigation_i}{\overline{w_{ij}}} \tag{3.35}
$$

It is not necessary to estimate the cost of capital because capital is measured in currency units (€).

Equation 3.34 can also be used to show how irrigation water users will respond to water price changes. When used to predict responses to water price changes, the policy water price, pw_{policy}, is substituted for c_{2ij}.

When the approach based on PMP is used, welfare changes are also measured by changes in net benefits. In this approach, production and input levels are estimated by solving the optimization problem presented in Equation 3.34, with the policy water price substituted for the observed water price. For each irrigation water use location, net benefits are estimated as follows:

$$NB_j = \sum_{i=1}^{N} \begin{pmatrix} \left(py_i + ps_{ij}\right) \cdot \alpha_{ij} \cdot (\beta_{1ij} \cdot x_{1ij}{}^{\gamma} + \beta_{2ij} \cdot x_{2ij}{}^{\gamma} + \beta_{3ij} \cdot x_{3ij}{}^{\gamma})^{1/\gamma} \\ -c\_land_{ij} \cdot x_{1ij} - p\_policy_w \cdot x_{2ij} - c_3 \cdot x_{3ij} \end{pmatrix} \tag{3.36}$$

This equation shows that land costs used in the net benefit calculation are based on land rental cost estimates and not on the quadratic land cost function estimated as part of the PMP approach. Although the PMP approach assumes that land costs are represented by a quadratic cost function, the methodology is actually representing a situation in which the land cost function is linear (i.e., land has a rental price) and constraints that are not observable in the baseline data set limit the extent to which different crops can be grown. The CES production functions and quadratic land cost functions are meant to capture these constraints, which may be due to land quality or to farmer expertise or other factors that are not observable but are probably "owned" by the landowner. In this case, the landowner then receives a profit as payment to these factors, which can be interpreted as a welfare gain. All profits above the costs of water, capital, and the opportunity cost of not renting land (i.e., the land rental cost) should be considered a welfare gain to the landowner. Therefore, changes in welfare estimated using the PMP approach should be measured by subtracting water costs, capital costs, and land rental opportunity costs from revenue.

Total net benefits at each location are measured as follows:

$$TNB_j = NB_j - c\_capital_j - c\_operating_j$$
$$-c\_pumping \cdot f\_gw_j \cdot \sum_{i=1}^{N} x_{2ij}$$

where (3.37)

$f\_gw_j =$ fraction of total water use from groundwater at location j in baseline simulation

It is assumed that the fraction of total water use from groundwater is the same as the fraction estimated in the baseline simulation.

3.6.3 Irrigation water use as a function of pumping costs

In the final investigation presented as part of this study, groundwater is priced using the price of energy as a surrogate for a water price. The Positive Mathematical Programming approach is used to predict how irrigation water users will respond to water price changes. Use of this approach requires a procedure for estimating how users will partition water use between surface water and groundwater when energy prices are used to control groundwater use. A procedure with four steps has been developed.

In the first step, it is assumed that water users have access to groundwater but not surface water. Profit-maximizing levels of groundwater use at each location are then estimated as follows:

$$\max \sum_{i=1}^{N_j} \left(py_i + ps_{ij} \right) \cdot \alpha_{ij} \cdot (\beta_{1ij} \cdot x_{1ij}{}^{\gamma} + \beta_{2ij} \cdot x_{2ij}{}^{\gamma} + \beta_{3ij} \cdot x_{3ij}{}^{\gamma})^{1/\gamma}$$

$$-a_{ij} \cdot x_{1ij} - 0.5 \cdot b_{ij} \cdot x_{1ij}{}^{2} - pe_{policy} \cdot \frac{\rho \cdot g}{\eta} \cdot x_{2ij} \cdot \left(-h_{\circ} + ad(Q_{ij}, L_{dij}) \right) - c_3 \cdot x_{3ij}$$

subject to

$$\sum_{i=1}^{N} x_{1ij} \le \sum_{i=1}^{N} A_{ij}$$

$$\sum_{i=1}^{N} x_{2ij} \le \sum_{i=1}^{N} w\_irrigation\_base_{ij}$$

where

$$Q_{ij} = \frac{x_{2ij}}{L_{pij} \cdot n_{ij}}$$

$L_{dij} =$ length of drawdown period for crop i at location j (d)

$L_{pij} =$ length of pumping period for crop i at location j (d)

$n_{ij} =$ number of wells providing water to crop i at location j (3.38)

The length of pumping period is assumed to equal 180 days for all crops at all locations. 180 days is used because this is the length of a typical irrigation season. It is further assumed that water levels in production wells will completely recover during the period of the year when irrigation does not take place, so the drawdown period is also equal to 180 days for all crops and locations. The number of wells is estimated using Equation 3.9. It is assumed that each crop at each location is served by one or more independent wells (i.e., no well provides water to more than one crop).

In the second step, marginal costs of water use are estimated for each crop and compared the price of surface water. If the marginal cost of groundwater for any crop exceeds the surface water price, then groundwater use by that crop is limited to an amount that makes the marginal cost of groundwater equal to the surface water price.

for $i = 1 : N_j$

$$mc_{2ij} = pe_{policy} \cdot \frac{\rho \cdot g}{\eta} \cdot \left(-h_{\circ} + 2 \cdot ad(Q_{ij}, L_{dij}) \right)$$

if $mc_{2ij} > pw_{policy}$

$$gw_{ij}^{*} = \frac{pw_{policy} - c}{2 \cdot d_{ij}} \qquad (3.39)$$

where

$mc_{2ij} =$ marginal cost of groundwater use by crop i at location j (€/m^3)

$gw_{ij}^{*} =$ maximum groundwater use by crop i at location j (m^3)

In the third step, the PMP optimiziation is re-run with the cost of water depending on whether the marginal cost of water determined in the first step is greater than the surface water price. If the marginal cost of water determined in the first step is greater than the surface water price, then the cost of water is computed using the volumetric surface water price. If the marginal cost of water determined in the first step is less than or equal to the surface water price, then the cost of water is computed using the energy price.

$$\max \sum_{i=1}^{N_j} \left(py_i + ps_{ij} \right) \cdot \alpha_{ij} \cdot (\beta_{1ij} \cdot x_{1ij}{}^{\gamma} + \beta_{2ij} \cdot x_{2ij}{}^{\gamma} + \beta_{3ij} \cdot x_{3ij}{}^{\gamma})^{1/\gamma}$$

$$-a_{ij} \cdot x_{1ij} - 0.5 \cdot b_{ij} \cdot x_{1ij}{}^{2} - c_{water}\left(x_{2ij} \right) - c_3 \cdot x_{3ij}$$

subject to

$$\sum_{i=1}^{N} x_{1ij} \le \sum_{i=1}^{N} A_{ij}$$

$$\sum_{i=1}^{N} x_{2ij} \le \sum_{i=1}^{N} w\_irrigation\_base_{ij}$$

where

$c_{water} = pw_{policy} \cdot x_{2ij}$ if $mc_{2ij}^{*} > pw_{policy}$

$c_{water} = pe_{policy} \cdot \frac{\rho \cdot g}{\eta} \cdot x_{2ij} \cdot \left(-h_{\circ} + ad(Q_{ij}, L_{dij}) \right)$ if $mc_{2ij}^{*} \le pw_{policy}$

$$mc_{2ij}^{*} = pe_{policy} \cdot \frac{\rho \cdot g}{\eta} \cdot \left(-h_{\circ} + 2 \cdot ad(Q_{ij}^{*}, L_{dij}) \right)$$

$$Q_{ij}^{*} = \frac{x_{2ij}^{*}}{L_{pij} \cdot n_{ij}} \text{ (m}^3\text{/d)} \qquad (3.40)$$

$x_{2ij}^{*} =$ level of annual water use identified in first step (m^3)

In the last step, water use is partioned between groundwater and surface water.

$$
\begin{aligned}
&\text{if } mc_{2ij}^{\ *} > pw_{policy} \\
&\quad gw_{ij} = gw_{ij}^{\ *} \\
&\quad sw_{ij} = x_{2ij} - gw_{ij} \\
&\text{else} \\
&\quad gw_{ij} = x_{2ij} \\
&\quad sw_{ij} = 0
\end{aligned}
\tag{3.41}
$$

where

$gw_{ij} =$ groundwater use by crop i at location j (m^3)

$sw_{ij} =$ surface water use by crop i at location j (m^3)

For the second pricing policy, net benefits are estimated as follows:

$$
\begin{aligned}
NB_j = &\sum_{i=1}^{N}\left(py_i + ps_{ij}\right)\cdot \alpha_{ij} \cdot (\beta_{1ij}\cdot x_{1ij}^{\ \gamma} + \beta_{2ij}\cdot x_{2ij}^{\ \gamma} + \beta_{3ij}\cdot x_{3ij}^{\ \gamma})^{1/\gamma} - c\_land_{ij}\cdot x_{1ij} \\
&- pw_{policy}\cdot sw_{ij} - pe_{policy}\cdot \frac{\rho\cdot g}{\eta}\cdot x_{2ij}\cdot\left(-h_{\circ} + ad(Q_{ij}, L_{dij})\right) - c_3\cdot x_{3ij}
\end{aligned}
\tag{3.42}
$$

In the net benefit calculation, the rental cost of land replaces the PMP quadratic land cost function. Total net benefits are estimated as follows:

$$
TNB_j = NB_j - c\_capital_j - c\_operating_j \tag{3.43}
$$

Groundwater pumping costs are not included in the total net benefit calculation because these costs are internalized in the groundwater costs paid by irrigation water users.

3.6.4 Urban/domestic water use

Urban/domestic water use is predicted assuming that urban/domestic water users can be represented as utility-maximizing consumers. Based on the principle of utility maximization, a demand function can be developed for a consumer that indicates how much of a good will be consumed as a function of price. In this study, a demand function is developed that indicates how much water a household will consume each month as a function of the price of water. This function is developed using the point-expansion method. For a derivation of the function, the reader is referred to Griffin (2006). The function has the following form:

$$w = k \cdot p_w^{\ e} \tag{3.44}$$

where

$w =$ monthly water use (m^3)

$k =$ demand function constant

$p_w =$ water price (€/m^3)

$e =$ price elasticity of water demand

The parameters of the demand function constant k are estimated using information from a baseline data set available for the case study basin. Volumetric water pricing is used at all locations in the case study basin.

$$k_j = \frac{w\_obs_j \cdot \left(1 - loss_j\right)}{p\_obs_j^{\ e}} \tag{3.45}$$

where

$j =$ urban/domestic water use location index

$k_j =$ demand function constant for location j

$w\_obs_j =$ total wholesale water use at location j in baseline data set (m^3/month)

$p\_obs_j =$ observed water price at location j in baseline data set (€/m^3)

$loss_j =$ observed loss fraction at location j in baseline data set

The price elasticity of demand is estimated to be -0.5 at all water use locations (for a meta-analysis of price elasticities, see Dalhuisen, 2003).

At each urban/domestic water use location, it is assumed that water is supplied to households by a water provider that sets water prices equal to marginal costs of water supply. Each urban/domestic water supply entity in the case study basin has access to groundwater but not to surface water. Under the first water pricing policy, the water supply entity's costs are the sum of volumetric water costs, groundwater pumping costs, and fixed operating costs. The water supply entity's cost function is as follows:

$$C\left(w\_domestic_j\right) = pw_{policy} \cdot w\_domestic_j + c\_pumping_j \cdot w\_domestic_j + c\_fixed_j$$

where

$C\left(w\_domestic_j\right) =$ cost of producing water at urban/domestic location j (€ / month) (3.46)

$w\_domestic_j =$ volume of water abstracted at location j (m^3/month)

$c\_pumping_j =$ pumping cost at location j in baseline data set (€/m^3)

$c\_fixed_j =$ fixed cost at location j in baseline data set (€/month)

It is assumed that the water provider operates at the point where marginal profits equal marginal costs.

$$pr_j \cdot \left(1 - loss_j\right) = pw_{policy} + c\_pumping_j$$

where (3.47)

$pr_j =$ retail water price at location j (€/m^3)

The equation presented above can be used to solve for the optimal water price.

$$pr_j = \frac{pw_{policy} - c\_pumping_j}{1 - loss_j} \tag{3.48}$$

This retail price can then be used to identify the amount of water that will be abstracted by the water provider.

$$w\_domestic_j = \frac{k_j \cdot pr_j^{\,e}}{1 - loss_j} \tag{3.49}$$

Welfare changes to consumers are estimated using the concept of consumers' surplus. Because water users would still consume water even if the price of water was higher, but probably in smaller amounts, the difference between willingness to pay and the actual price paid is interpreted as a "welfare gain" to the water user, called the consumer's surplus. The concept of consumer's surplus is illustrated graphically in Figure 3.6. The figure shows a representative consumer demand curve indicating how water use by a representative consumer varies as a function of water price. The rectangular area at the bottom of the figure represents the actual amount paid for consuming a given amount of water.

The area below the demand curve and above the rectangle representing the cost of water is the consumer's surplus.

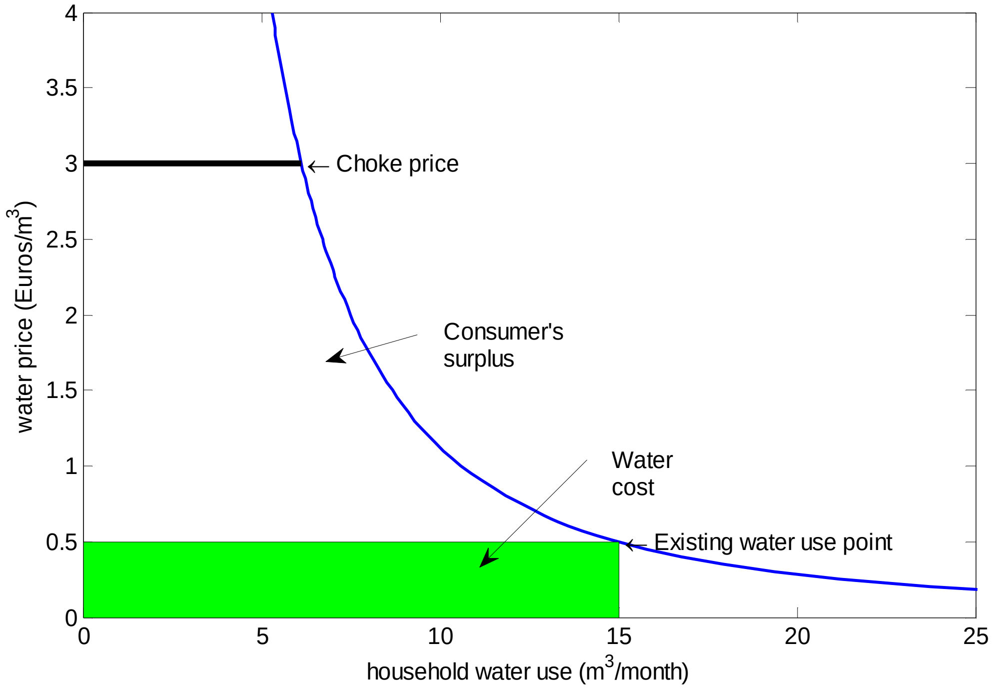

Figure 3.6: Consumers' surplus

The consumers' surplus is capped by a "choke price" representing the point at which consumers would switch to an alternative water source. In a water valuation study in Sydney, Australia, Grafton and Ward (2008) suggest that the choke price be based on the average cost per cubic meter of constructing a rainwater collection tank. This approach was applied here with the assumption that rainwater can substitute for marginal household water uses such as landscaping and cleaning. The cost of installing rainwater collection facilities seems more appropriate for the choke price than other candidates such as bottled water, as it seems unlikely that households would purchase bottled water for cleaning and landscaping.

In the modelling framework, consumers do not switch to rainwater collection as water prices increase because prices do not reach the level of the choke price. The purpose of the choke price is to cap the consumer demand curve so consumer's surplus can be calculated; otherwise, in the constant elasticity functional form used here, consumer willingness to pay approaches infinity as water use approaches zero.

Consumers' surplus is measured as follows:

$$cs_j = N_{months} \cdot \int_{pr_j}^{p\_choke} k_j \cdot p^e dp$$

where

$cs_j =$ annual consumers' surplus at location j (€/year) (3.50)

$N_{months} = 12$

$p\_choke =$ choke price (€/m^3)

It is also necessary to measure profits to urban/domestic water suppliers.

$$profit_j = N_{months} \cdot \begin{pmatrix} pr_j \cdot w\_domestic_j \cdot (1 - loss_j) - pw_{policy} \cdot w\_domestic_j \\ -c\_pumping_j \cdot w\_domestic_j - c\_fixed_j \end{pmatrix}$$

where (3.51)

$profit_j =$ annual profit to supplier at location j (€)

The total net benefit at each is location is the sum of the consumers' surplus and the water supplier's profit.

$$NB_j = cs_j + profit_j$$

where (3.52)

$NB_j =$ net benefit at location j (€)

3.6.5 Urban/domestic water use as a function of pumping costs

In the third investigation presented as part of this study, groundwater is priced using an energy price as a surrogate for a water price. At each urban/domestic water use location, it is assumed that water is supplied to households by an entity that operates so that marginal benefits of operation are equal to marginal costs. The water supplier's costs are the sum of groundwater pumping costs and fixed operating costs.

$$C\left(w\_domestic_j\right) = pe_{policy} \cdot \frac{\rho \cdot g}{\eta} \cdot w\_domestic_j \cdot \left(-h_\circ + ad(Q_j, L_{dj})\right)$$
$$+c\_fixed_j$$
where
$$Q_j = \frac{w\_domestic_j}{L_{pj} \cdot n_j} \tag{3.53}$$
$L_{dj} =$ length of drawdown period at location j (d)

$L_{pj} =$ length of pumping period at location j (d)

$n_j =$ number of wells providing water at location j

The length of puming period in is assumed to equal 365 days at all locations because it is assumed that pumping takes place year-round. The drawdown period for urban/domestic water users is estimated to be 20 years. Because the purpose of this study is to predict responses to water price changes in the long term, a long time horizon is used to estimate drawdown. A period of 20 years is chosen because the simulation period for the hydrological model of the case study basin is 20 years. This model is used to predict whether groundwater pumping rates predicted as a result of water price changes are sustainable in the long term, as was outlined in section 3. The drawdown period is assumed to be equal to the length of the simulation period of the hydrological model so that estimates of drawdown and estimates of groundwater sustainability are developed using a consistent time scale. The number of wells is estimated using Equation 3.9.

The water supplier is assumed to operate at the point where marginal profits equal marginal costs:

$$pr_j \cdot \left(1 - loss_j\right) = pe_{policy} \cdot \frac{\rho \cdot g}{\eta} \cdot \left(-h_\circ + 2 \cdot ad(Q_{ij}, L_{dij})\right) \tag{3.54}$$

This equation can be solved to identify the amount of water that will be produced as a function of the retail water price.

$$w\_domestic_j = \frac{pr_j \cdot \left(1 - loss_j\right) - pe_{policy} \cdot \frac{\rho \cdot g}{\eta} \cdot -h_{\circ}}{2 \cdot pe_{policy} \cdot \frac{\rho \cdot g}{\eta} \cdot \frac{1}{\frac{n_j \cdot L_{pj}}{4 \cdot \pi \cdot T}} \cdot \left(\ln\left(\frac{2.25 \cdot T \cdot L_{dj}}{r^2 \cdot S} \right) - 1 \right)} \tag{3.55}$$

A retail water price that equates supply and demand can then be found by setting the supply function equal to the household water demand function.

$$\frac{pr_j \cdot \left(1 - loss_j\right) - pe_{policy} \cdot \frac{\rho \cdot g}{\eta} \cdot -h_{\circ}}{2 \cdot pe_{policy} \cdot \frac{\rho \cdot g}{\eta} \cdot \frac{1}{\frac{n_j \cdot L_{pj}}{4 \cdot \pi \cdot T}} \cdot \left(\ln\left(\frac{2.25 \cdot T \cdot L_{dj}}{r^2 \cdot S} \right) - 1 \right)} = \frac{k_j \cdot pr_j^{\,e}}{1 - loss_j} \tag{3.56}$$

If this equation is solved for the retail water price, pr_j, then it is possible to identify a monthly water use volume using Equation 3.55.

Supplier profits are estimated as follows:

$$profit_j = N_{months} \cdot \left(pr_j \cdot w\_domestic_j \cdot \left(1 - loss_j\right) - C\left(w\_domestic_j\right)\right) \tag{3.57}$$

Total net benefits at each location are then equal to the sum of supplier profits and consumers’ surplus.

3.6.6 Industry water use

Industry water use is estimated by estimating a maximum willingness to pay for water for each industy and comparing this to the average cost of water supply. If an industry’s maximum willingness to pay for water is greater than the average cost of water supply, it is assumed that the industry will use water at the level identified as part of the baseline data set. If the average cost of water supply exceeds maximum willingness to pay, it is assumed that the industry goes out of production and uses no water.

The maximum willingness to pay for each industry is estimated using the residual imputation method. Estimates of input costs for industry water users are not available as part of the baseline data set. Input costs are estimated using a data set obtained from the Greek Statistics office (Hellenic Statistical Authority, 2008). This data set has estimates of revenues, labor costs, and non-labor input costs for different regions in Greece. These data have been aggregated by industry type. It is assumed that the ratio of total input costs to revenues for industries in the baseline data set can be estimated using these data. The ratio of total input costs to revenues for each industry is assumed to equal the ratio of aggregated input costs to aggregated revenues for that industry's industry type and region. Maximum willingness to pay is then estimated as follows:

$$WTP_{ij} = \frac{\left(1 - \dfrac{input_{kl}}{revenue_{kl}}\right) \cdot turnover_{ij}}{w\_industry\_base_{ij}}$$

where

$WTP_{ij} =$ maximum willingness to pay for water for industry i at location j (€/m^3)

$i =$ industry index

$j =$ industry water use location index

$k =$ industry type index

$l =$ region index

where

$\dfrac{input_{kl}}{revenue_{kl}} =$ ratio of input costs (€/year) to revenues (€/year) for industry type k in region l

$turnover_{ij} =$ turnover for industry i at location j in baseline data set (€)

$w\_industry\_base_{ij} =$ annual water use by industry i at location j in baseline data set (m^3) (3.58)

industry i $\in$ industry type k

industry water use location j $\in$ region l

The method presented above estimates a maximum willingness-to-pay value that is probably higher than the true value. This is because of the difficulty of including all input costs apart from water in the willingness-to-pay calculation.

Water supply costs are equal to the sum of volumetric water costs and groundwater pumping costs. The cost function is as follows:

$$
\begin{aligned}
&C\left(w\_industry_{ij}\right) = pw_{policy} \cdot w\_industry_{ij} + c\_pumping_j \cdot w\_industry_{ij} \\
&\text{where} \\
&i = \text{ industry index} \\
&j = \text{ location index} \\
&C\left(w\_industry_{ij}\right) = \text{ cost of producing water for industry i at location j (€ / year)} \\
&w\_industry_{ij} = \text{ water use by industry i at location j (m}^3\text{/year)} \\
&c\_pumping_j = \text{ pumping cost at location j in baseline data set (€/m}^3\text{)}
\end{aligned}
\tag{3.59}
$$

Industry water use is then predicted as follows:

$$
\begin{aligned}
&\text{if } WTP_{ij} \geq pw_{policy} + c\_pumping_j \text{ then} \\
&\quad w\_industry_{ij} = w\_industry\_base_{ij} \\
&\text{else} \\
&\quad w\_industry_{ij} = 0 \\
&\text{end}
\end{aligned}
\tag{3.60}
$$

Net benefits are then measured as follows:

$$
\begin{aligned}
&NB_j = \sum_{i=1}^{N} w\_industry_{ij} \cdot \left(WTP_{ij} - pw_{policy} - c\_pumping_j\right) \\
&\text{where} \\
&NB_j = \text{ annual net benefit at location j (€/year)} \\
&N = \text{ number of industries at location j}
\end{aligned}
\tag{3.61}
$$

3.6.7 Industry water use as a function of pumping costs

In the final investigation, groundwater is priced using an energy price as a surrogate for a water price. In this case, the industry cost function as follows:

$$C\left(w\_industry_{ij}\right) = pe_{policy} \cdot \frac{\rho \cdot g}{\eta} \cdot w\_industry_{ij} \cdot \left(-h_{\circ} + ad(Q_{ij}, L_{dij})\right)$$

where

$$Q_{ij} = \frac{w\_industry_{ij}}{L_{pij} \cdot n_{ij}} \tag{3.62}$$

L_{dij} = length of drawdown period for industry i at location j (d)

L_{pij} = length of pumping period for industry i at location j (d)

n_{ij} = number of wells providing water to industry i at location j

The length of pumping period varies depending on whether industries operate seasonally or year-round. If an industry operates seasonally, the pumping period is equal the number of days that the industry operates per year. If the industry operates year-round, the pumping period is 365 days.

The drawdown period also varies depending on whether industries are seasonal or year-round. In the case of seasonal industries, it is assumed that well heads recover during periods when industries are not operating. For industries that operate year-round, the drawdown period is assumed to equal 20 years.

The number of wells is estimated using Equation 3.9. It is assumed that each industry at each location is served by one or more independent wells (i.e., no well provides water to more than one industry).

Industry water use is then predicted using the following procedure:

if $WTP_{ij} \geq ac_{ij}$ then

$\quad w\_industry_{ij} = w\_industry\_base_{ij}$

else

$\quad w\_industry_{ij} = 0$ (3.63)

end

where

$$ac_{ij} = pe_{policy} \cdot \frac{\rho \cdot g}{\eta} \cdot \left(-h_{\circ} + ad(Q_{ij}, L_{dij})\right)$$

Average groundwater pumping costs are used in Equation 3.63 instead of marginal costs. The maximum willingness to pay value is an estimate of the upper limit of the average value of water use observed in the baseline data set.

This value may not be equal to the marginal value of water. The average value of water is equal to the marginal value when production is a linear function of water, but this is unlikely to be the case for any of the industries in the baseline data set. Therefore, the maximum willingness to pay should not be used as a guide to identifying a profit-maximizing level of pumping, as would be implied if marginal costs of groundwater pumping were used in Equation 3.63. Instead, the maximum willingness to pay is interpreted as an upper limit on the cost of groundwater pumping.

Net benefits are measured as follows:

$$NB_j = \sum_{i=1}^{N_j} WTP_{ij} \cdot w\_industry_{ij} - pe_{policy} \cdot \frac{\rho \cdot g}{\eta} \cdot w\_industry_{ij} \cdot \left(-h_{\circ} + ad(Q_{ij}, L_{dij})\right) \quad (3.64)$$

3.7 Optimization approach

An optimization approach is used to identify an appropriate water price, or combination of a water price and energy price. The optimization approach is presented for the case of a single water price in this section. The second approach, in which optimization is used to identify appropriate water and energy prices, is described in the results section. In the first approach, where optimization is used to find a single volumetric water price, the optimization problem is described using the following objective function:

$$\max \sum_{j=1}^{N} NB_j$$

where

$j =$ water use location index (3.65)

$N =$ number of water use locations in case study basin

$NB_j =$ annual net benefit at location j (€)

The decision variable in the optimization problem above is the water price. The objective function is subject to the following constraints:

$\sigma_1 \le \sigma_{\text{target}}$

$\sigma_2 \le 0.1$ (3.66)

$\sigma_{\text{target}} =$ ecological status target

Different values of the ecological status target are used in this simulation. The impact of different values on model outcomes is reported in the results section.

The optimization approach is implemented using the following steps:

1. A water price is selected.
2. Annual or monthly levels of water use are identified for all water users in the basin using the approaches described in section 3.5
3. Annual net benefits are estimated for all water users based on levels of water use identified in step 2.
4. Annual or monthly levels of water use identified in step 2 are converted into daily timeseries of water demands for the 20-year simulation period of the MIKE BASIN hydrological model.
5. The hydrological model is run for the 20-year simulation period with the demands identified in step 4 as boundary conditions.
6. After the hydrological model run is complete, the values of the ecological status and groundwater sustainability parameters are computed
7. A new water price is selected and steps 2-6 are repeated.

Steps 1-7 are repeated until a water price is found that maximizes net benefits at the basin scale subject to the ecological status and groundwater sustainability constraints. A gradient search optimizer available as part of the Matlab software package (Matlab fmincon) is used to find the optimal water price.

Step 4 of the optimization approach requires that annual or monthly water use levels be converted into 20-year timeseries of daily water demands. Daily timeseries of irrigation water use demands are identified using different procedures for each water use type. The procedure used to identify daily demands for irrigation water users depends on whether the residual impuation or Positive Mathematical Programming approach is being used.

If the residual imputation method is being used, no daily water demands are estimated. Instead, crop areas identified using Equation 3.15 are written to the MIKE BASIN model. Daily crop water use is then estimated dynamically by the MIKE BASIN irrigation model.

If the Positive Mathematical Programming approach is being used, a two-step procedure is used to convert annual water use levels to daily demands. In the first step, the annual water use level predicted by the PMP approach is divided by the baseline annual water use level to determine the fraction of the baseline water use level that will be used. In the second step, daily surface water and groundwater use timeseries values from the baseline simulation are multiplied by this fraction. In the baseline simulation data set, it is possible to observe total water use by individual crop type. However, it is not possible to observe how much of the total is from surface water and how much is from groundwater. Information on the partitioning of total water use between surface water and groundwater is only available at the irrigation water user level. Therefore, timeseries estimates of groundwater and surface water use at the crop level are estimated by multiplying total water use observed at the crop level by groundwater and surface water use fractions observed at the irrigation node level. The procedure is summarized below:

$$sw_{ijk} = r_{ij} \cdot f\_sw_{jk} \cdot w\_irrigation\_base_{ijk}$$
$$gw_{ijk} = r_{ij} \cdot f\_gw_{jk} \cdot w\_irrigation\_base_{ijk}$$

where

$k =$ day index

$sw_{ijk} =$ surface water use by crop i at location j during day k (m^3)

$gw_{ijk} =$ surface water use by crop i at location j during day k (m^3)

$$r_{ij} = \frac{x_{2ij}}{w\_irrigation\_base_{ij}}$$

$f\_sw_{jk} =$ fraction of surface water use at location j during day k in baseline simulation

$f\_gw_{jk} =$ fraction of groundwater use at location j during day k in baseline simulation

$w\_irrigation\_base_{ijk} =$ water use by crop i at location j during day k in baseline simulation (m^3) (3.67)

Domestic water demands are identified using the following procedure:

$$w\_domestic_{jk} = \frac{w\_domestic_j}{\text{number of days per month}}$$

where (3.68)

$w\_domestic_{jk} =$ water demand at location j during day k (m^3)

It is assumed that monthly demands are constant over the entire simulation period.

Industry demands are identified using the following procedure:

$$w\_industry_{jk} = \frac{w\_industry_k}{\text{days that industry } k \text{ is active per year}}$$

(during days that industry j is active)

$w\_industry_{jk} = 0$ (during days that industry j is inactive) (3.69)

where

$w\_industry_{jk} =$ demand by industry j during day k (m^3)

Annual water demands by industry are assumed to be constant over the 20-year simulation period. Some industries are seasonal and therefor have seasonal water demands

This approach simplifies the optimization problem because a single decision variable can be used to optimize water use over the entire analysis area. However, it is also possible to use spatially and/or temporally varying water prices, which would of course increase the number of decision variables. Indeed, Pulido-Velazquez et al. (2008) found that shadow prices of environmental flow constraints are higher during dry periods, signalling the increased value of water during periods of water scarcity, while Cai (2008) found that marginal values of water use can vary within a catchment area. The use of a monthly or seasonal water price, for example, might signal the increased value of water in the driest months of the summer irrigation season and motivate a switch to crops that are less sensitive to irrigation during dry months. A spatially varying water price could also be used to shift consumptive use to subcatchments with smaller impacts on basin flow patterns. In this analysis, a single price has been used both to keep the optimization problem simple and because it seems that it may be difficult for a river basin authority to implement more complex pricing schemes.

3.8 Uncertainty analysis

The author is not aware of a comprehensive treatment of uncertainty in a hydro-economic modelling study. The hydro-economic modelling approach presented here presents challenges for uncertainty analysis because it is being used to predict an outcome to a situation (volumetric water pricing) that is not observable, which means that no calibration data set is available. A sensitivity analysis is still useful in this situation, but it seems that such an analysis should consider the simultaneous impact of different sources of uncertainty. In a model with many uncertain boundary conditions, parameters, and structures, it is challenging to quantify the extent to which different boundary conditions, parameters, and structures contribute to uncertainty observed in model results. In addition, it is necessary to have information about the probability distributions of model inputs in order make statistical inferences about the impact of these inputs on model results (for a state-of-the-art review of sensitivity analysis, see Saltelli et al, 2006). It is particularly difficult to estimate probability distributions for the economic parameters used in this study because this implies knowledge of future economic conditions, which are highly uncertain.

The uncertainty approach used in this paper is based on the Info-Gap decision analysis framework, which was introduced by Ben-Haim (2004). The Info-Gap framework was developed as an alternative to probabilistic approaches for evaluating model uncertainty. In the Info-Gap framework, some model boundary conditions, parameters, or structures are assumed to be subject to uncertainty that can not be described using probability distributions. These boundary conditions, parameters, and structures are varied simulataneously over increasingly larger ranges to estimate the extent to which resulting model outcomes are sensitive to uncertainty. After sensitivity to increasing uncertainty has been estimated, guidelines are suggested for using this information in the decision-making process. The Info-Gap approach was selected because it provides a method for considering the simultaneous impact of different sources of uncertainty on model outcomes without specifying probability distributions for these different sources of uncertainty. In addition, it provides useful guidelines for using this information in a decision-making process. The method is straightforward to apply and has reasonable computational requirements.

The Info-Gap decision analysis approach centers around two concepts: robustness and opportuneness. A robust decision is defined as one that maximizes the extent to which uncertainty can be introduced before the minimum value of a model prediction fails to meet a critical standard. An opportune decision is defined as one that minimizes the extent to which uncertainty can be introduced before a model predicts a value above windfall value that is desired but not required.

In this case, the decision to be analyzed is the selection of a water price. The robustness and opportuneness of the water price decision are estimated with respect to the objective of achieving the ecological status target.

With regard to the ecological status target, the Info-Gap robustness function can be written as follows:

$$
\begin{aligned}
&\max\{a : \sigma_{1*}[pw_{policy}, \mathbb{F}(a, \tilde{f})] \leq \sigma_{\text{target}}\}, a \geq 0 \\
&\text{where} \\
&a = \text{ Info-gap uncertainty parameter} \\
&\sigma_{1*} = \text{ maximum value of ecological status parameter} \\
&\text{associated with uncertainty level } a \\
&\mathbb{F}(a, \tilde{f}) = \text{ set of model outcomes} \\
&\text{associated with uncertainty level } a \\
&\tilde{f} = \text{ nominal model outcome}
\end{aligned}
\tag{3.70}
$$

In the equation above, the parameter a is used to define a horizon of uncertainty around the nominal outcome $\tilde{f}$. The nominal outcome is the result of running the complete model framework described here for a given policy water price. For each value of a, there is a set of model outcomes, $\mathbb{F}(a, \tilde{f})$ all which differ from the nominal outcome by an amount that is less than the horizon of uncertainty defined by a. In each set, there is a value σ_{1*} that defines the highest (worst) value in the set of model outcomes. The objective of the robustness function is to maximize the horizon of uncertainty that can exist before at least one of the model outcomes in the set $\mathbb{F}(a, \tilde{f})$ associated with that level of uncertainty has a value of σ_{1*} that exceeds the ecological status target.

With regard to the ecological status target, the Info-Gap opportuneness function can be written as follows:

$$\min\{a : \sigma_1^*[pw_{policy}, \mathbb{F}(a, \tilde{f})] \le \sigma_{\text{desired}}\}, a \ge 0$$

where

$\sigma_1^* =$ minimum value of ecological status parameter associated with uncertainty level a (3.71)

$\sigma_{\text{desired}} =$ value of ecological status parameter that is desired but not required

In each set $\mathbb{F}(a, \tilde{f})$, there is a value σ_1^* that defines the lowest (best) value in the set of model outcomes. The objective of the opportuneness function is to minimize the horizon of uncertainty that can exist before at least one of the model outcomes in the set $\mathbb{F}(a, \tilde{f})$ associated with that level of uncertainty has a value of σ_1^* that is lower than a value of the ecological status parameter that is desired but not required.

Different approaches are possible for describing the horizon of uncertainty associated with each value of the Info-Gap uncertainty parameter a. This analysis uses an approach based on Regan et al. (2005). This approach is described as follows:

1. Select value of a
2. Allow model parameters to vary within the range

$$x \cdot \max((1-a), 0) \le x \le x \cdot (1+a)$$

where

$x =$ nominal model parameter value (3.72)

$a =$ info-gap uncertainty parameter (same value used for all model inputs)

3. Sample model parameter values from a uniform probability distribution bounded by the range defined in step 2 and compute results (i.e., average annual net benefits of water use) using water price identified during optimization run.
4. Repeat using different sets of random samples until a representative set of model outcomes is obtained.
5. Increase value of a and repeat.

Once a representative set of model outcomes has been identified for each uncertainty level, these outcomes can be used to estimate the robustness or opportuneness of different decisions.

As defined in Equation 3.72, the info-gap uncertainty parameter a can be interpreted as a maximum percentage change in input values. For example, if the value of the info-gap uncertainty parameter is 0.5, then each input value is sampled from a uniform distribution that is bounded on the lower end by a value equal to 50% of the nominal value and on the upper end by a value equal to 150% of the nominal value. To develop a model outcome associated with a value of a, input values are sampled randomly from the range defined by a and the resulting model outcome is computed. This process is repeated until a representative sample set has been developed. The value of a is then increased and a new set is developed. Although parameter values are sampled assuming a uniform distribution, this does not imply any assumptions about the likelihoods of different values; rather, the uniform distribution is used as a tool to ensure that all possible values are sampled so that that combinations of parameter values with the most significant impacts on model results are identified.

Once the sets of possible model outcomes associated with the different values of a have been identified, the information can be used for decision-making using the concepts of robustness and opportuneness. Using the decision-making application described here, a water price is considered robust if the value of a can be increased signficantly without having any of the possible model outcomes in the set associated with that value of a violate the ecological status target. A water price is considered opportune if a small increase in the value of a results in at least one model outcome in the set associated with that value of a that is less than a lower value of the ecological parameter that is desired but not required.

The hydro-economic model presented here is sensitive to uncertain economic and hydrologic data, as well as uncertainty regarding the model structure. The uncertainty analysis approach is limited to uncertain economic data. Because the purpose of the study is to estimate the average annual impact of a policy change, the analysis assumes that the range of hydrological conditions found in the 20-year simulation period contains enough variability to capture hydrological uncertainties. Uncertainty in the model structure is not addressed here.

4 Overview of main results

Three investigations are undertaken using the approach presented in section 3.

In the first investigation, the approach is used to identify a water price that will result in an ecological status parameter value of 0.25. A single volumetric water price is applied to both surface water and groundwater. The residual imputation method is used to predict how irrigation water users will respond to water price changes. The Info-Gap uncertainty analysis approach is used to estimate the impact of uncertainty on model results. This investigation is described in paper I.

In the second investigation, the residual imputation method is compared to the Positive Mathematical Programming approach. In this investigation, a single volumetric water price is applied to both surface water and groundwater. The purpose of the investigation is to compare how the two approaches predict that irrigation water users will respond to water price changes. This investigation is described in paper II.

In the final investigation, the optimization approach is used to identify a set of water prices that will result in an ecological status parameter value of 0.5. The set of water prices consists of a volumetric surface water price and an energy price that is used to control groundwater use. The Positive Mathematical Programming approach is used to predict how irrigation water users will respond to water price changes. This investigation is described in paper III.

4.1 Investigation 1

In the first investigation, the optimization approach is used to identify a single volumetric water price that results in an ecological status parameter value of 0.25. This water price is 1.06 €/m^3. The distribution of net benefits and water use among water use types that results from this water price is shown in Figure 4.1.

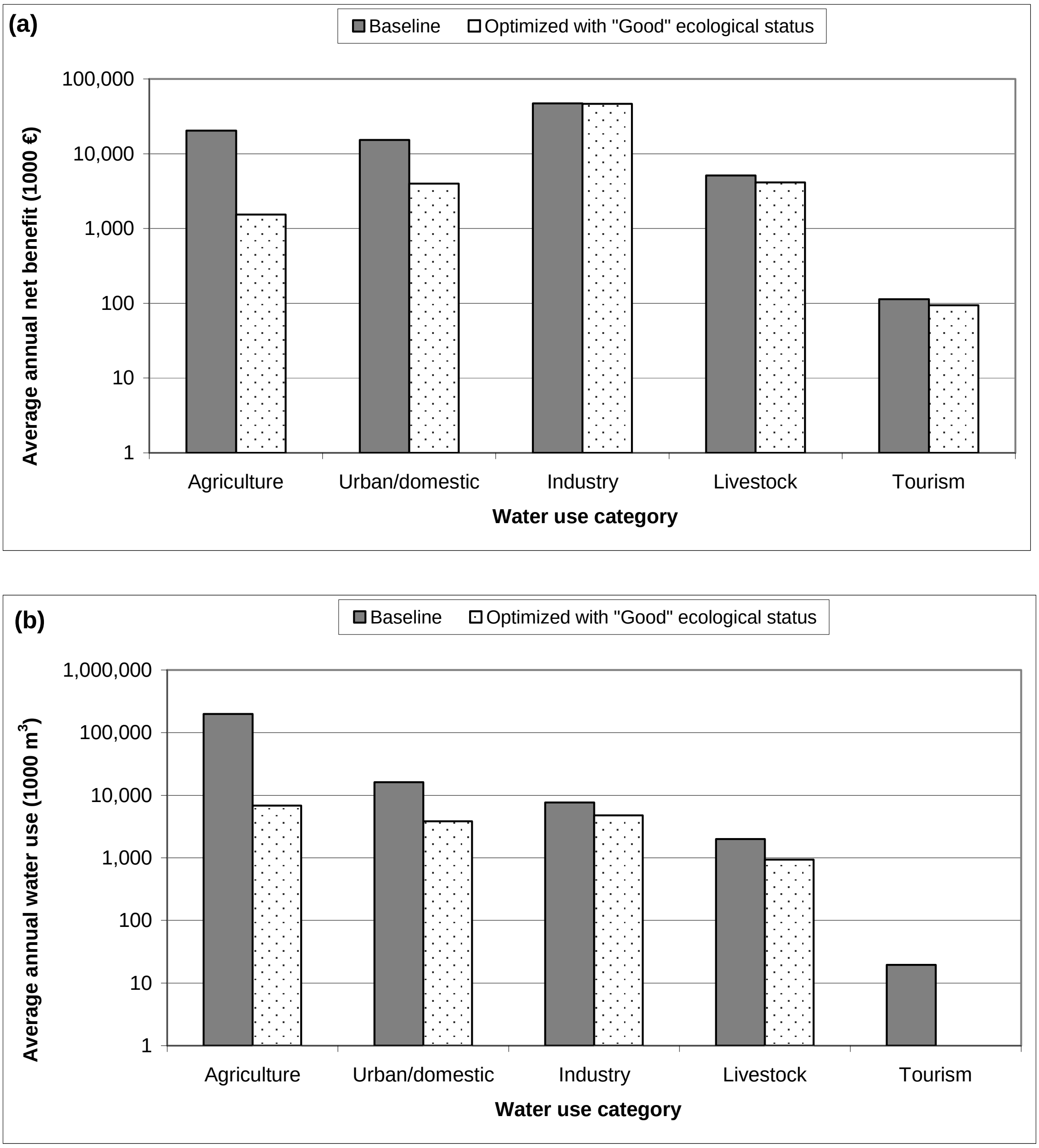

Figure 4.1: Comparison of (a) net benefits and (b) water use. The columns labeled "Optimized with 'Good' ecological status" describe results observed when the price of water is 1.06 €/m^3.

Figure 4.1 suggests that majority of opportunity costs appear to be borne by the agriculture sector, which could lose as much as 90% of the value attributed to agricultural water use in the baseline scenario. Significant costs are also borne by the urban/domestic sector. The overwhelming majority of water use cutbacks occur in the agriculture sector.

In the first investigation, the approach described in section 3 is modified so that urban/domestic, industry, livestock, and tourism water users have the option to switch to desalinated water supplies if these supplies are less expensive than natural water from the river and groundwater network. For most users, the price of developing and using desalinated supplies is between 1 €/m^3 and 1.50 €/m^3. Therefore, when the price of natural water is set to 1.06 €/m^3, some of these users switch to desalinated supplies. Methods used to estimate costs of developing desalinated supplies are presented in paper I.

The impact of a water price increase on ecological status is shown in Figure 4.2, which compares daily flow histories for three scenarios: unmodified "natural" flows, a baseline representing existing water use, and a scenario where the water price is equal to 1.06 €/m^3. A comparison of cumulative flow distributions for the month of August is also shown. The figure shows that the flow history of the scenario in which the water price is equal to 1.06 €/m^3 matches the unmodified flow scenario more closely than the baseline scenario, particularly in the summer months. This is confirmed by the plot comparing cumulative flow distributions for the month of August.

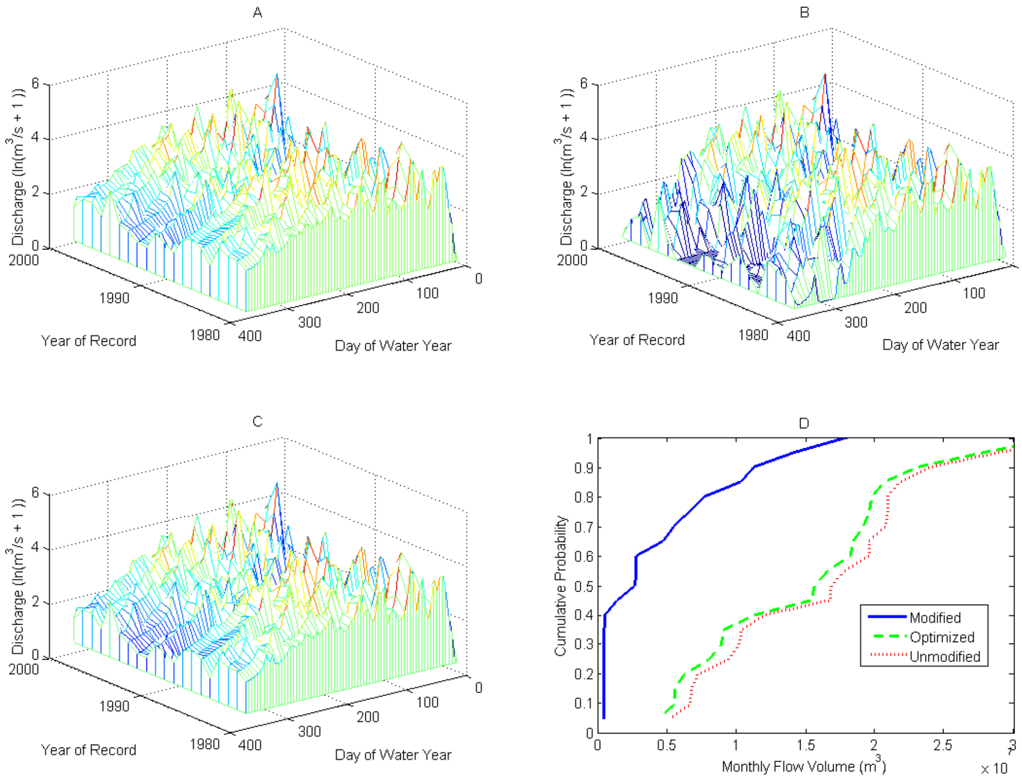

Figure 4.2: Comparison of flow histories at basin outlet. (A) Unmodified scenario. (B) Baseline scenario. (C) Water price = 1.06 €/m^3. (D) Comparison of August CDFs. "Optimized" CDF refers to scenario in which water price = 1.06 €/m^3.

The impact of uncertainty on model predictions is estimated using the robustness and opportuneness concepts that are part of the Info-Gap framework. Figure 4.3 plots maximum and minimum values of the ecological status parameter for the set associated with each value of the Info-Gap uncertainty parameter a. The line labeled "robustness" gives maximum values and the line labeled "opportuneness" gives minimum values. The plot shows how these values change as economic input parameters are allowed to vary by up to ±80% from nominal values. The plot indicates that a water price of 1.06 €/m^3 is not robust if the ecological status target is 0.25. However, if an ecological status target of 0.4 were to be adopted, then it appears that implementing a water price of 1.06 €/m^3 would be a robust decision that would be likely to meet the target. On the other hand, it appears unlikely that an ecological status target of less that 0.2 could be achieved.

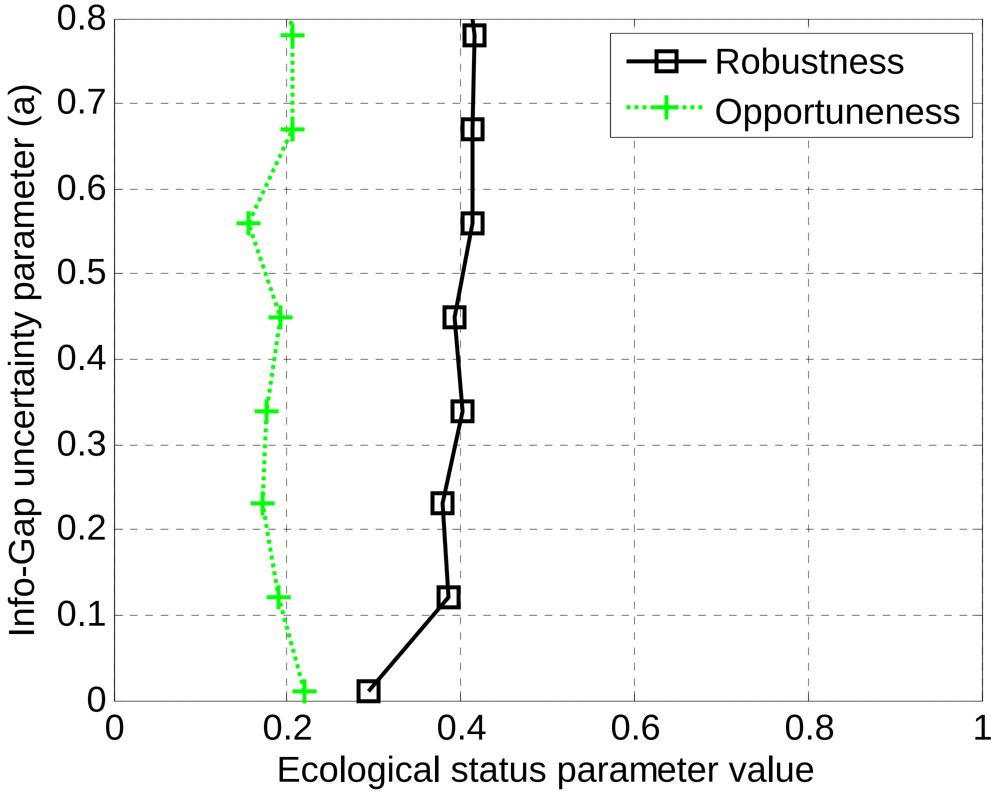

Figure 4.3: Info-Gap results. Each point on the robustness line is the maximum value of the ecological status parameter observed in the set of values associated with a value of the Info-Gap uncertainty parameter. Each point on the opportuneness line is the minimum value of the ecological status parameter observed in the set values associated with a value of the Info-Gap uncertainty parameter.

Other water prices besides the "optimal" water price were evaluated to estimate how these decisions might perform under uncertainty. Figure 4.4 plots robustness and opportuneness for four water prices that are higher than the "optimal" water price. The figure suggests that there may be few benefits to be gained from using a water price higher than the "optimal" water price. Water prices of 1.16 €/m^3, 1.26 €/m^3, and 1.36 €/m^3 would not be robust unless the ecological status target was changed to 0.4. A water price of 1.46 €/m^3 would be somewhat robust if the ecological status target was changed to around 0.3. None of the higher prices seem to increase the opportuneness significantly; in other words, none of the prices increase that likelihood that an ecological status value significantly lower than the 0.25 target will occur.

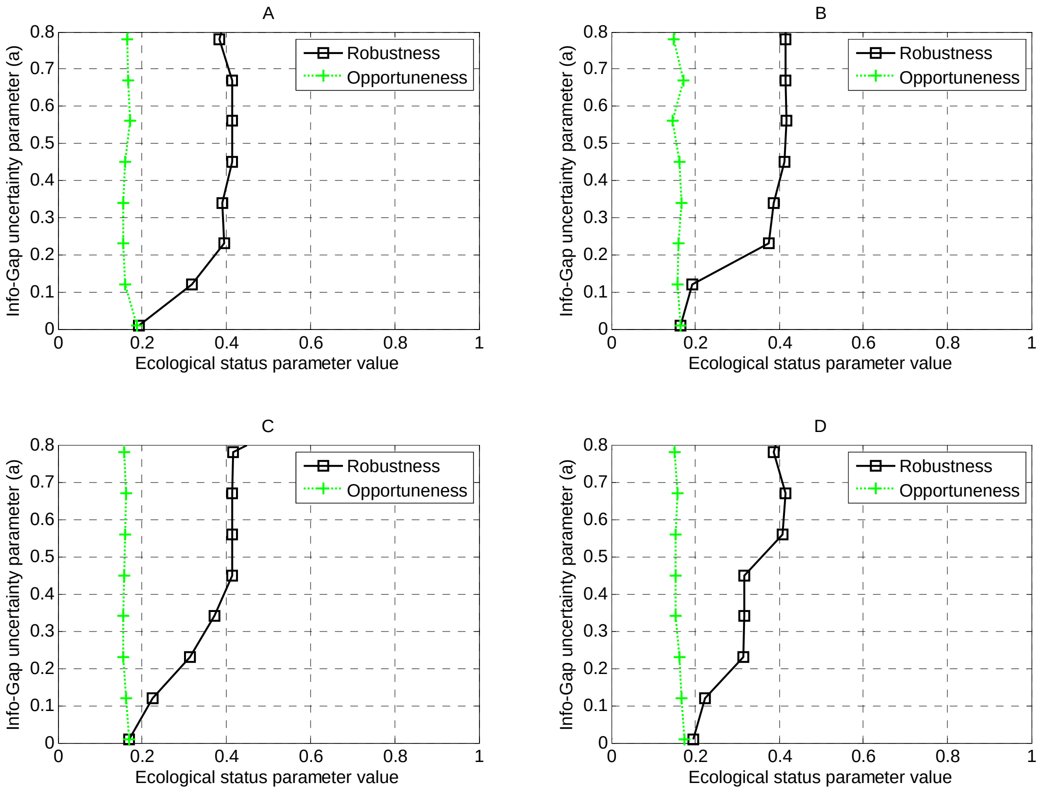

Figure 4.4: Info-Gap results. A: Water price = 1.16 €/m^3. B: Water price = 1.26 €/m^3. C: Water price = 1.36 €/m^3. D: Water price = 1.46 €/m^3.

Figure 4.5 plots robustness and opportuneness for four water prices that are lower than the "optimal" water price. This figure suggests that if an ecological status target of 0.4 is selected, then lower water prices would be robust with respect to this target. A water price of 0.66 €/m^3, which is the lowest of the four prices presented in the figure, is still robust to uncertainty if the target is 0.4 However, the figures show that opportuneness decreases as the water price is reduced. Although a water price of 0.96 €/m^3 and a water price of 0.66 €/m^3 are equally robust with respect to a target of 0.4, a price of 0.96 €/m^3 is more opportune with respect to the original target of 0.25. In other words, at a higher water price, less uncertainty must be introduced before it is possible that the 0.25 target will be met.

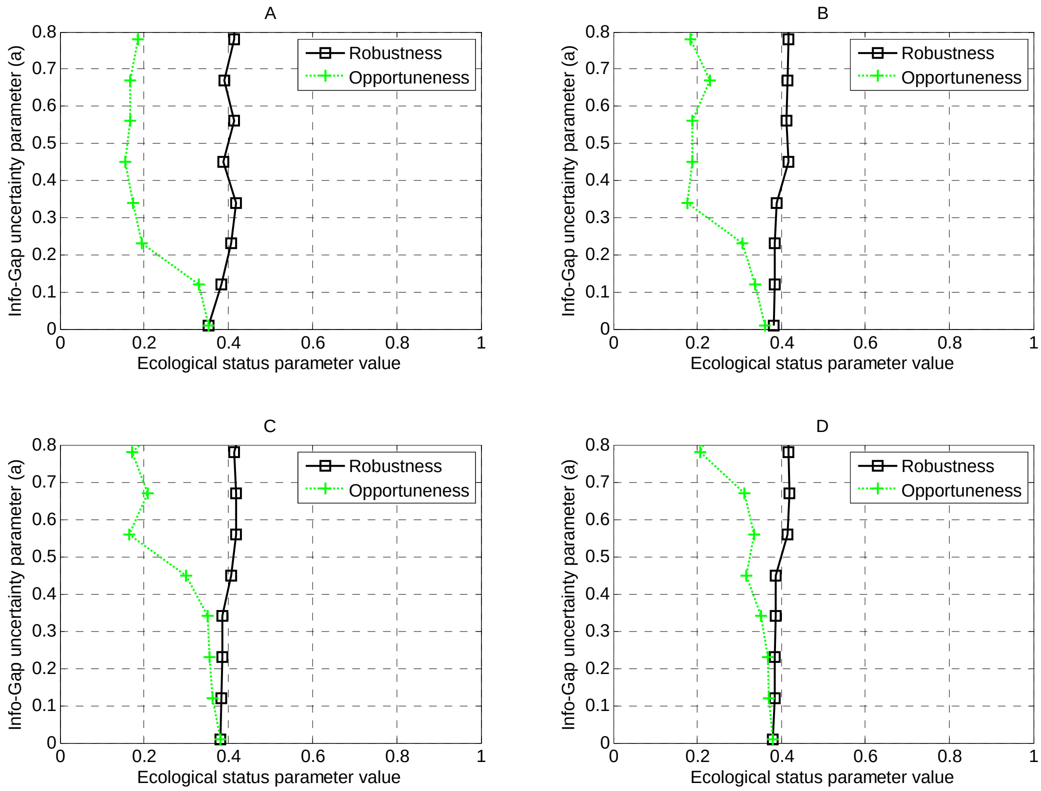

Figure 4.5: Info-Gap results. A: Water price = 0.96 €/m^3. B: Water price = 0.86 €/m^3. C: Water price = 0.76 €/m^3. D: Water price = 0.66 €/m^3.

4.2 Investigation 2

In the second investigation, the residual imputation method is compared to the Positive Mathematical Programming approach. A comparison of crop areas predicted by the two approaches is presented in Figure 4.6 The figure compares crop areas over a range of water prices for one irrigation water use location. Similar results are observable at other water use locations.

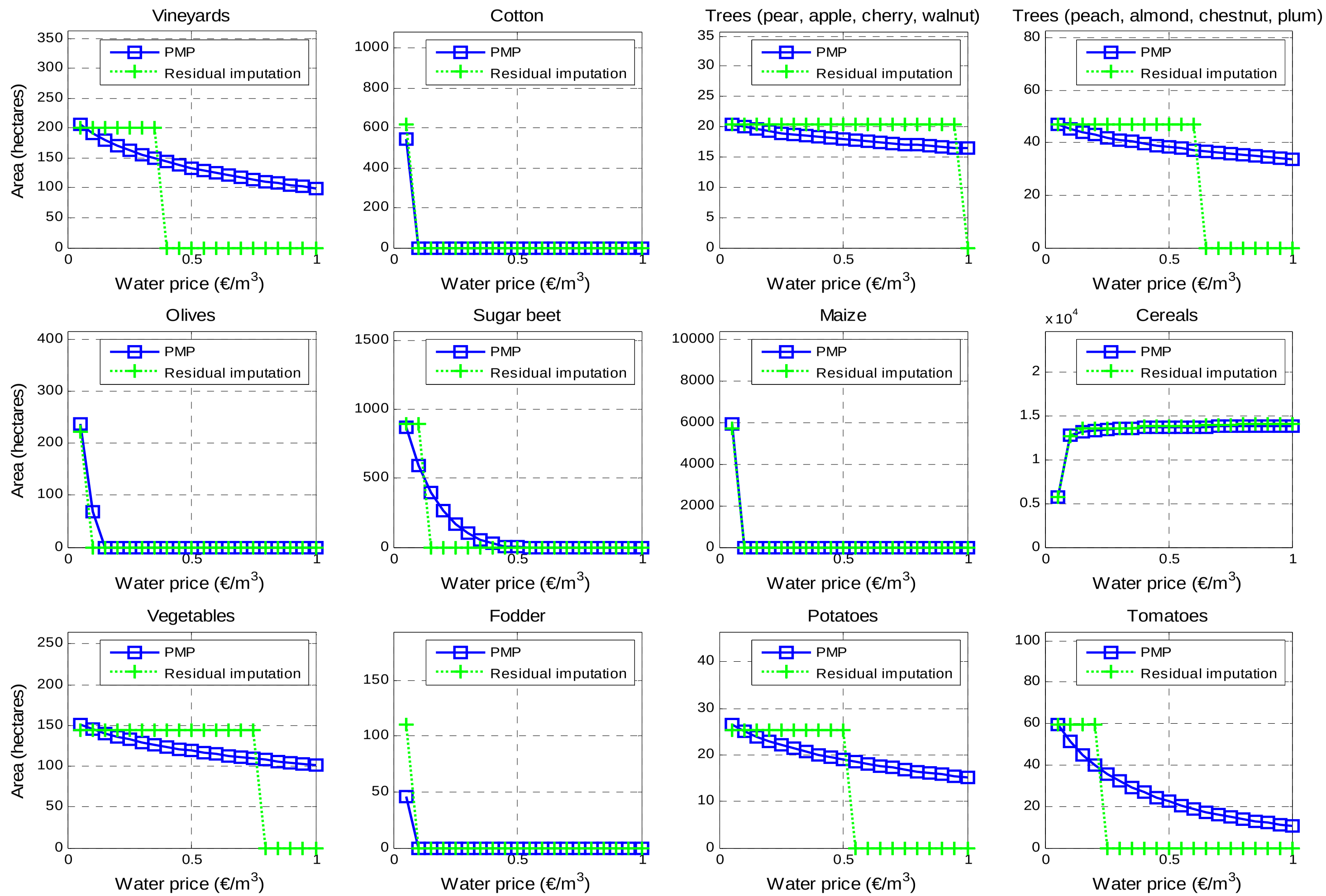

Figure 4.6: Comparison of crop areas predicted by PMP and residual imputation approaches, irrigation water use location 375.

Figure 4.6 shows that the PMP approach predicts more gradual changes in land use than the binary changes predicted by the residual imputation approach. However, it is interesting to note that the two approaches make similar predictions for some crops, including cotton, olives, maize, fodder, and cereals. The PMP approach predicts it is not profit maximizing to grow cotton, olives, maize, and fodder at higher water prices. For each of these crops, the price at which the PMP approach predicts that they will no longer be grown is similar to the marginal value estimated using the residual method. This is why the two approaches predict that that these crops will go out of production at similar water price levels. cereals are the only non-irrigated crop included in this study, which is why the cereals area is observed to increase as the water price increases.

The PMP and residual imputation approaches also predict that water use will change as a function of increasing water prices. Figure 4.7 presents unit water use as a function of water price for a number of crop types. The figure presents data from irrigation water use location 375, but the same trends are observable at other water use locations. The figure shows that the PMP approach predicts that it is profit-maximizing to use a deficit irrigation strategy for a number of crops, including Vineyards, orchard crops, Tobacco, Vegetables, Potatoes, and Tomatoes.

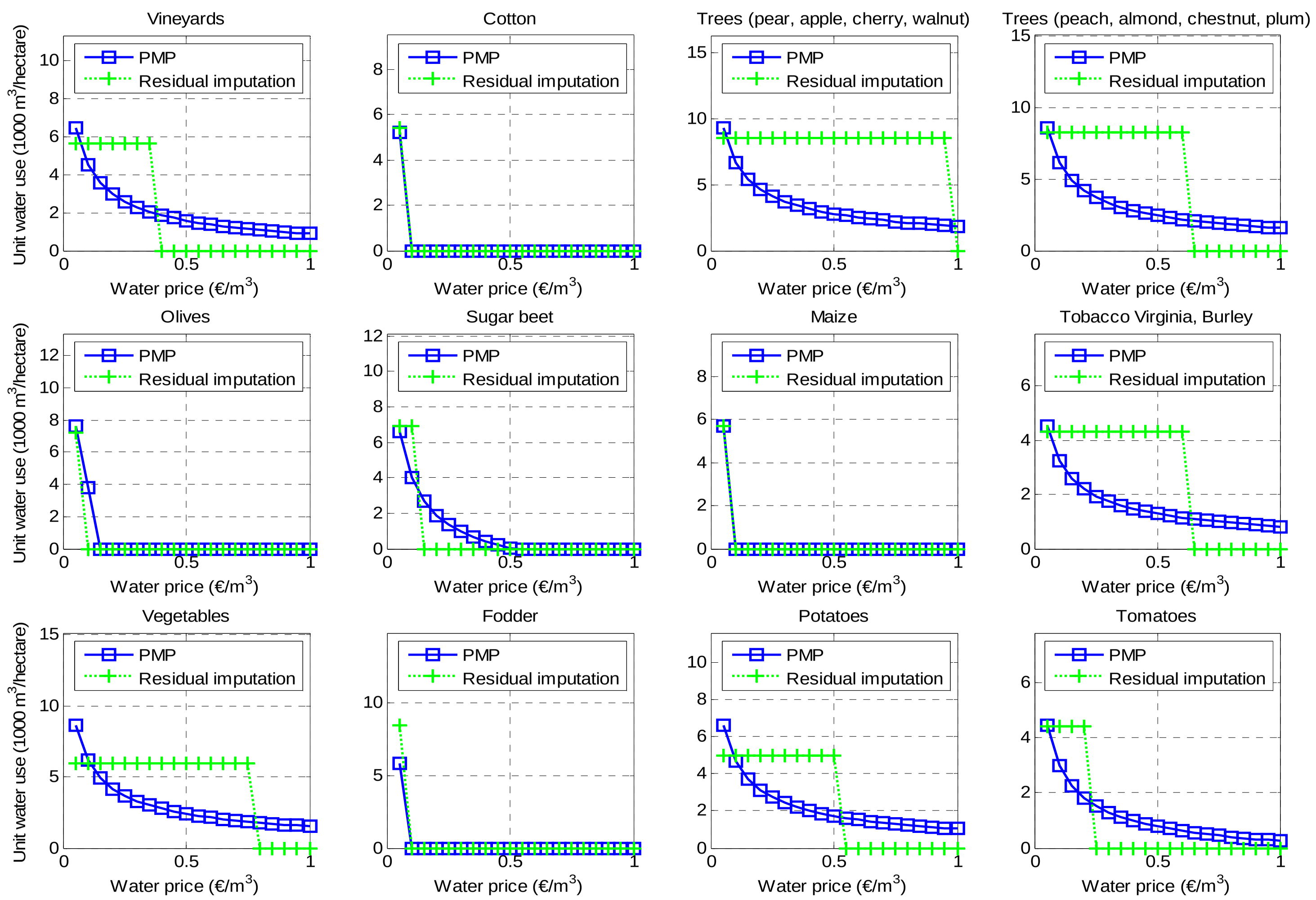

Figure 4.7: Comparison of unit water use predicted by PMP and residual imputation approaches at irrigation water use location 375.

Because of the deficit irrigation strategies observable in Figure 4.7, the PMP approach predicts that more irrigated crops will remain in production at higher water prices. Figure 4.8 presents total irrigated and non-irrigated land use predicted by the two approaches as a function of water price. The figure shows that the PMP approach predicts that some irrigated land will remain in production throughout the range of water prices considered in the study. However, both approaches predict that the overwhelming majority of irrigated lands will be converted to dryland agriculture.

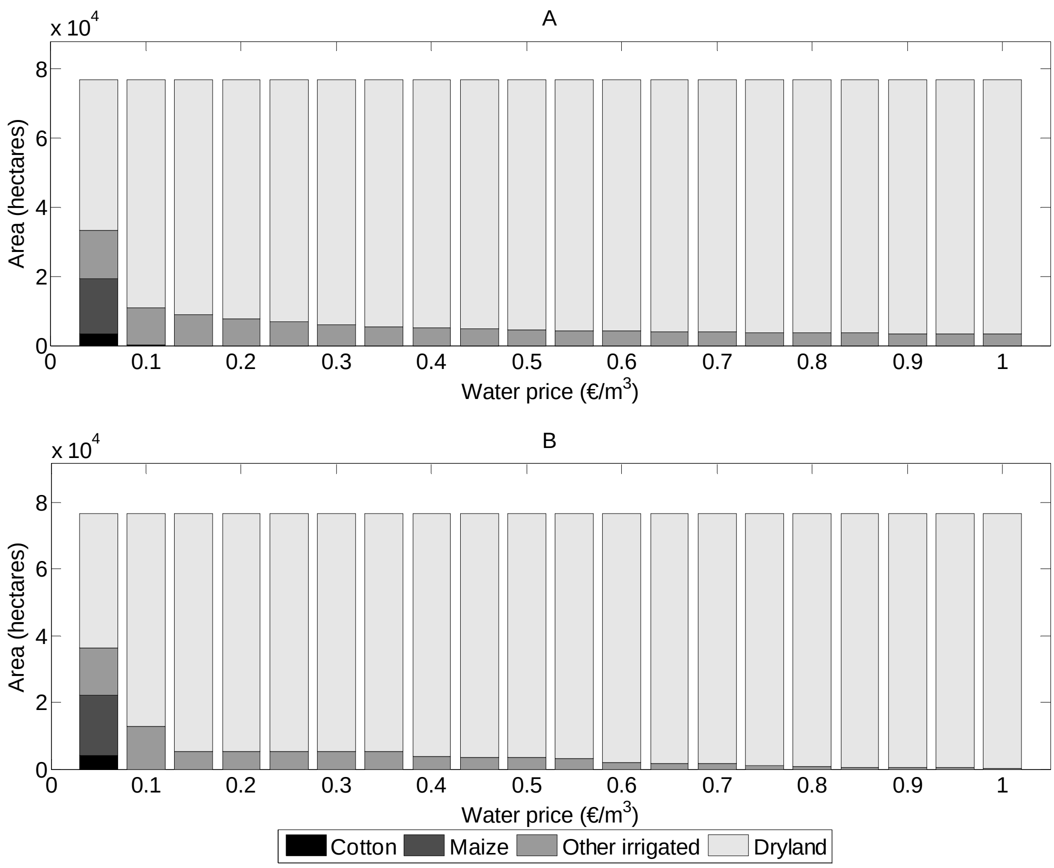

Figure 4.8: Comparison of land use predicted by (A) PMP and (B) residual imputation approaches at basin scale.

Although the PMP approach predicts that more land will stay in production at higher water prices, the approach also predicts that water use will decline more steeply than predicted by the residual imputation approach. Figure 4.9 compares annual water use predicted by the two approaches as a function of water prices. The figure shows that, for most water prices, the PMP approach predicts that annual water use at the basin scale will be less than predicted by the residual imputation approach. This is because the PMP approach predicts that it will be profit-maximizing to use deficit irrigation for many crops, while the residual imputation approach can not predict that deficit irrigation will take place.

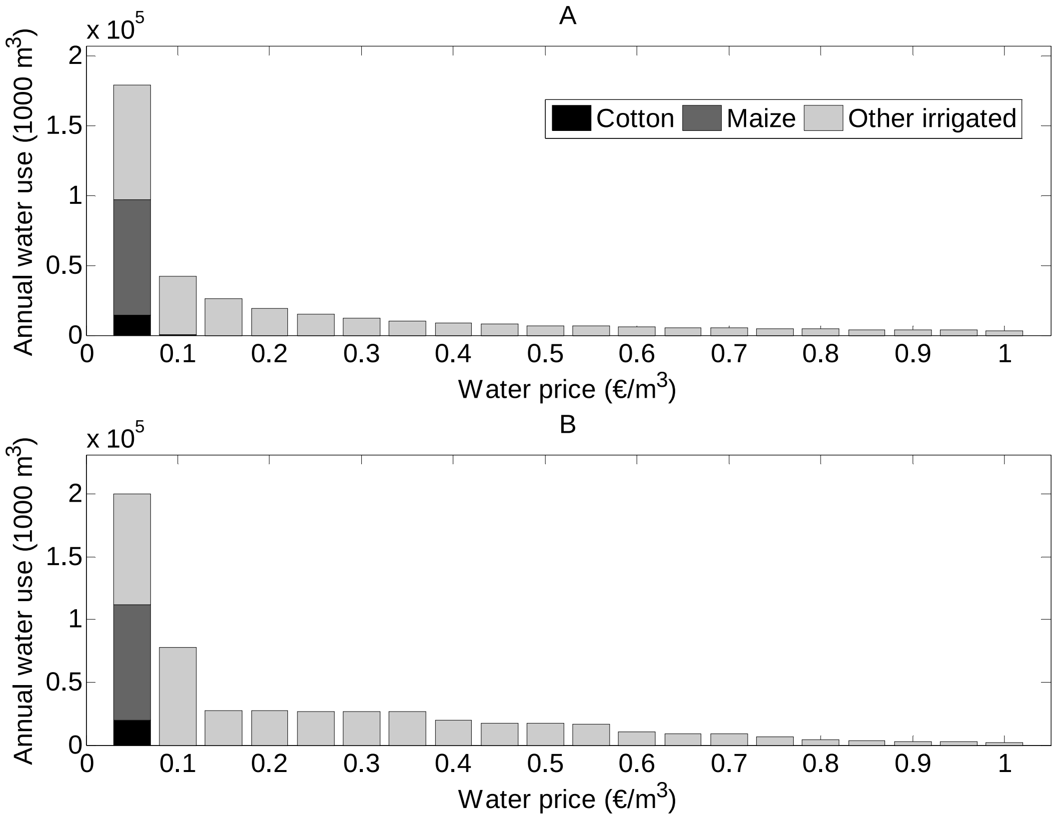

Figure 4.9: Comparison of annual water use at basin scale predicted by (A) PMP and (B) residual imputation approaches.

4.3 Investigation 3

In the final investigation, the optimization approach is used to identify a set of water prices that will result in an ecological status parameter value of 0.5. The set of water prices consists of a volumetric surface water price and an energy price that is used to control groundwater use. An energy price is used to control groundwater use by controlling the cost of groundwater pumping. Results are compared to results obtained using a single uniform price for both surface water and groundwater.

Optimal prices identified for each of the pricing policies are presented in Table 4.1. The table also presents estimates of basin-wide net benefits, ecological status, and the maximum depth to the deep groundwater layer. The values are compared to values estimated using the baseline data set.

Table 4.1: Comparison of optimal water prices and main indicators.

Parameter	Baseline	Pricing policy 1	Pricing policy 2
Optimal water price (€/m^3)	not applied	0.10	0.12
Optimal energy price (€/kWh)	not applied	not applied	1.13
Basin-wide net benefit (1000 €)	8.12E+04	6.85E+04	6.91E+04
Ecological status parameter value	0.89	0.44	0.47
Maximum depth to deep groundwater layer (m)	23.798	20.07	20.10

Table 4.1 shows that the optimal surface water price under the second water pricing policy is higher than the optimal water price identified under the first water pricing policy. Table 4.2 presents crop areas and water use predicted at one irrigation water use location when the first policy's price and the second policy's surface water price are both equal to 0.10 €/m^3. The energy price is set to the optimal price: 1.13 €/kWh. The table shows that a number of crops predicted to go out of production under the first pricing policy are still active under the second policy. These crops include cotton, maize, and fodder. The table also shows that total water use is higher under the second policy.

Total water use is higher under the second policy when volumetric prices are equal because the marginal cost of using groundwater is a function of the amount of water used. For this reason, it is possible that the marginal cost of using groundwater can be less than the surface water price. Crops such as cotton and maize are predicted to be active under the second pricing policy because it is profit-maximizing to continue to grow these crops if groundwater is available at lower marginal costs. Because total water use is higher, the surface water price must be increased to 0.12 €/m^3 under the second water pricing policy to reduce

Table 4.2: Comparison of land and water use at irrigation water use location 375.

	Baseline		First pricing policy, water price=0.10		Second water pricing policy, water price=0.10 €/m^3, energy price=1.13 €/m^3			
Crop	Land use (ha)	Water use (1000 m^3)	Land use (ha)	Water use (1000 m^3)	Land use (ha)	Groundwater use (1000 m^3)	Surface water use (1000 m^3)	Total water use (1000 m^3)
Vineyards	202	1136	192	866	192	179	687	866
Cotton	620	3363	0	0	59	219	0	219
Trees (pear, apple, cherry, walnut)	20	173	20	134	20	146	0	146
Trees (peach, almond, chestnut, plum)	47	387	46	280	46	179	100	280
Olives	222	1607	69	260	69	179	81	260
Sugar beet	899	6236	592	2359	592	717	1642	2359
Maize	5740	32751	0	0	228	1301	0	1301
Tobacco Virginia, Burley	36	157	35	114	36	131	0	131
Vegetables	145	858	145	893	145	179	714	893
Fodder	111	937	0	0	19	92	0	92
Pulses	3	11	2	8	3	17	0	17
Potatoes	25	127	25	117	26	135	0	135
Cereals	5752	0	12792	0	12484	0	0	0
Tomatoes	60	263	51	153	52	164	0	164
Clover	223	1977	135	590	135	179	411	590
Totals	14104	49983	14104	5774	14104	3820	3635	7454

total water use to the point that ecological and groundwater sustainability constraints are satisfied.

Table 4.3 compares profits estimated for different crop types at one location in the model. Profits presented under the column heading "Pricing policy 2" represent profits estimated when water and energy prices are equal to the prices presented in Table 4.1 The table indicates that profits are higher under the first pricing policy for high value crops such as Vineyards and Vegetables. Under the second water pricing policy, profits are higher for low value crops like maize and cotton. This is a direct consequence of what was observed in Table 4.1 and Table 4.2. The introduction of energy prices to control groundwater allows some production of crops with low marginal water productivity because these crops are still profitable when the marginal cost of groundwater is low. This means that surface water prices must be increased to limit overall water use, which has the effect of reducing water use by crops with high marginal water productivity.

Table 4.3: Comparison of crop profits at irrigation water use location 375.

	Profit (1000 €)		
Crop	**Baseline**	**Pricing policy 1**	**Pricing policy 2**
Vineyards	393	309	296
Cotton	168	0	6
Trees (pear, apple, cherry, walnut)	166	152	160
Trees (peach, almond, chestnut, plum)	240	207	210
Olives	100	0	6
Sugar beet	588	102	94
Maize	1347	0	32
Tobacco Virginia, Burley	93	80	87
Vegetables	604	585	571
Fodder	46	0	2
Pulses	5	5	6
Potatoes	61	55	63
Cereals	300	667	657
Tomatoes	47	25	32
Clover	177	20	19
Total	4336	2207	2242

Net benefits are distributed differently in the urban/domestic sector under the two pricing policies. This can be explained by the retail water prices estimated under both policies, which are presented in Table 4.4. The table shows that retail prices estimated under the second policy are more variable than prices observed under the first policy. Retail water prices predicted under the second policy are positively correlated with water use. Higher retail water prices are estimated in locations with high water use because high water use increases groundwater pumping and drawdown.

Table 4.4: Comparison of retail water prices for urban/domestic users.

Water use location	Number of connections, baseline	Water use per connection, baseline (m^3/month)	Retail water price, baseline (€/m^3)	Retail water price, pricing policy 1 (€/m^3)	Retail water price, pricing policy 2 (€/m^3)
286	1005	16.2	0.45	0.17	0.14
287	1921	18.8	0.43	0.17	0.23
288	1662	17.1	0.45	0.17	0.20
289	2188	17.6	0.45	0.19	0.26
290	23963	24.0	0.45	0.33	1.02
291	3273	22.9	0.28	0.28	0.43
292	1916	22.5	0.28	0.26	0.29
293	3167	31.7	0.43	0.28	0.60
294	1530	21.3	0.28	0.23	0.23
296	2175	19.9	0.30	0.19	0.24
297	2613	18.1	0.45	0.19	0.29
298	4550	22.9	0.28	0.28	0.53
322	236	16.8	0.45	0.17	0.07
323	2126	19.9	0.30	0.19	0.24
326	443	32.8	0.43	0.28	0.19
327	1013	32.8	0.43	0.28	0.30
328	1337	22.9	0.28	0.28	0.25
330	423	21.3	0.28	0.23	0.11
331	521	21.3	0.28	0.23	0.12

The impact of the two pricing policies on ecological status can be observed in Figure 4.10. Flow history plots are presented for four scenarios: the baseline, an unmodified flow scenario, the first pricing policy at the optimal water price (0.10 €/m^3), and the second pricing policy at optimal prices (0.12 €/m^3 and 1.13 €/kWh). The figure indicates that both pricing policies result in a flow regime that resembles the unmodified flow regime more closely than the flow regime simulated using the baseline data set. In particular, it can be seen that both pricing policies predict that flows will continue throughout the summer irrigation season, when flows can be extremely low if baseline water use is assumed.

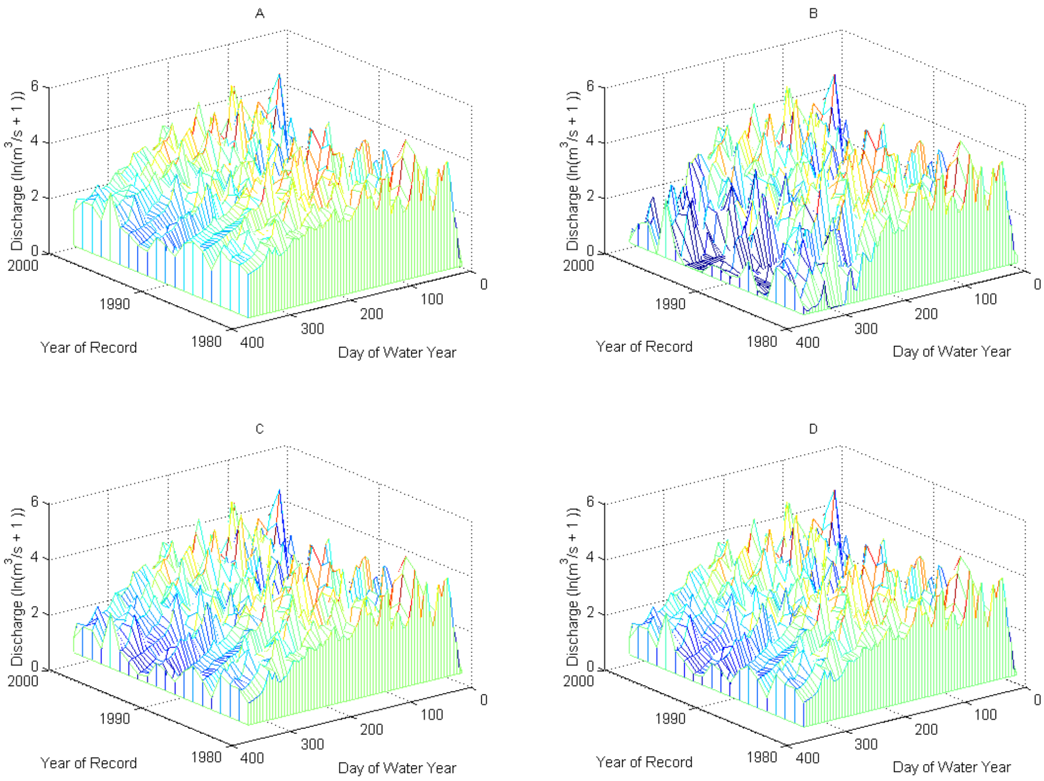

Figure 10: Comparison of flow history plots. A) Unmodified flow scenario. B) Baseline scenario. C) First water pricing policy (water price = 0.10 €/m^3). D) Second water pricing policy (water price = 0.12 €/m^3, energy price = 1.13 €/kWh).

5 Discussion

The approach presented here implements the ecological status and groundwater sustainability objectives of the WFD by eliminating water uses with low marginal values. Under the assumptions used in this analysis, the water uses with lowest marginal values are mostly agricultural water uses. Agriculture is also the dominant water user in the catchment area. The majority of resource or opportunity costs are therefore borne by the agriculture sector. This result is not unexpected or unusual, as agriculture often has a significant impact on water resources, through consumptive use, water quality impacts, or both.

Viewed from another perspective, it could be argued that the agricultural sector causes an environmental loss to society. This highlights the importance of including values of environmental goods in this kind of study. If an ecological status target can only be achieved with significant opportunity costs, an effort should be made to quantify environmental benefits gained from reaching the target using a common metric. Particularly in situations where the implementation of WFD requirements may be derogated on grounds of disproportionate costs, it seems important that the environmental benefits to be gained be included in the cost-benefit analysis.

This analysis does not consider potential feedbacks between changes in supply and changes in output prices. If agricultural production is reduced significantly because of a higher price on water, as this analysis assumes, this will also lead to a reduction in the supply of food products. As a consequence, the prices of agricultural products might be expected to increase. This again means that more agricultural production would be profitable even if the water price has increased. These kinds of feedbacks can only be analyzed using an equilibrium approach.

The Info-Gap decision theory is useful for uncertainty analysis in hydro-economic modelling. Hydro-economic modelling requires the use of many different boundary conditions, parameters, and model structures that are subject to uncertainty and may not be possible to represent probabilistically. For example, it seems difficult to use a probabilistic representation of future prices of agricultural commodities; even if one were developed using historical data, it seems plausible that circumstances could arise in the future that impact the prices of these commodities in new ways. Many different uncertain inputs are used in a hydro-economic modelling study and these inputs may interact to impact results. The Info-Gap approach provides a straightforward method to assess the simultaneous impact of uncertainty from different inputs on model outcomes when the probability distributions of these inputs are unknown.

The Info-Gap approach is also useful because it provides guidelines for how to use information about uncertainty in a decision-making process. Figures 4.3 to 4.5 suggest that information provided by the Info-Gap approach could be used in two decisions: the selection of a water price and the selection of reasonable target value for the ecological status parameter. Figure 4.3, which plots robustness and opportuneness for the "optimal" water price from the first investigation, indicates that this water price is not robust to uncertainty unless the ecological status parameter value is increased from 0.25 to 0.4. Robustness plots for higher water prices, which are presented in Figure 4.4, indicate that increasing the price of water does not increase robustness with respect to an original target value of 0.25; these plots also suggest that higher prices would be robust with respect to a target value of 0.4. Figure 4.5, which plots robustness and opportuneness for prices lower than the "optimal" water price from the first investigation, indicates that these lower prices are all robust with respect to a target value of 0.4. All of this suggests that 0.4 may be a more reasonable target value for the ecological status parameter than 0.25. In addition, Figure 4.5 shows that opportuneness with respect to lower values of the ecological statue parameter decreases as the price of water decreases. This suggests that although the lowest price (0.66 €/m^3) is robust with regard to a value of 0.4, a higher price could be implemented to increase the likelihood of achieving the original target of 0.25.

Under the assumptions used in this study, the PMP and residual imputation approaches make similar predictions about how irrigation water users will respond to water price changes. The two approaches make similar predictions even though the PMP approach is more flexible. The PMP approach can simulate deficit irrigation, substitution of other inputs for water, and changes in land use. The residual imputation method can not model deficit irrigation or substitution, and can only simulate the conversion of irrigated land to dryland crops.

Despite these differences, the two approaches make similar predictions because of the way in which production and land cost function parameters are estimated using the PMP approach. These parameters are estimated so that the production and cost functions will reproduce existing production and input levels without constraints if profit maximization is assumed. These parameters are strongly influenced by production and input levels observed in the baseline data set and limit the extent to which input levels and production are predicted to change when water prices are increased. At most of the irrigation water use locations in the case study area, most irrigated crop lands are allocated to low value crops including maize, cotton, Sugar beet, and fodder crops. Both the PMP approach and the residual imputation approaches predict that these crops will go out of production at fairly low water prices, and the prices at which the PMP approach predicts that it will no longer be profit-maximizing to grow these crops are similar to willingness to pay values identified using the residual imputation approach. Because the PMP production and land cost function parameters limit the extent to which high-value crops such as vegetables and orchard crops can then replace these low-value crops, the two approaches predict similar responses to water price changes.

Although the PMP approach does not predict that high-value irrigated crops will replace low value irrigated crops, the approach does predict that the production of high-value crops will continue to be profit-maximizing at higher water prices. For these crops, the PMP approach also predicts that deficit irrigation will be used. In this case, this prediction is based on economic criteria; because the values of these crops are high, they are still profitable when yields are reduced by a reduced water supply. However, it is interesting to note that this conclusion is supported by agronomic research into the use of deficit irrigation strategies for high-value crops such as fruit and vine crops. Both Fereres and Soriano (2007) and Ruiz-Sanchez et al. (2010) found that deficit irrigation can be a profitable strategy when applied to fruit and vine crops.

Are predictions made by the PMP approach reasonable? It seems contradictory to predict that deficit irrigation of high-value crops is profit-maximizing and at the same time to predict that the areas of these crops will not increase in response to water price changes. However, the PMP approach is meant to capture hidden constraints affecting crop production that are not observable in the baseline data set. These hidden constraints can include constraints related to land quality, market uncertainty, or management ability, among other factors. In the case of high-value crops, there is evidence that all three of the aforementioned factors limit the extent to which these crops are grown. Young (2005) has written an excellent discussion of why the area dedicated to high-value crops in irrigated regions is often less than would seem to be justified by the apparent profitability of these crops. Among other factors, high value crops are often perishable and vulnerable to weather and disease; are subject to significant producer price changes; require significant inputs of labor, fertilizer, and management expertise; and need high-quality soils. Given these constraints and uncertainties, it is not obvious that it would be possible to convert large amounts of land in to the production of high value crops. Indeed, some of the MCDM approaches cited in the literature review use labor minimization and risk minimization as criteria affecting farmer behavior in addition to profit maximization; it seems that both of these criteria would limit the extent of high-value crop production.

The predictions made by the PMP approach are also affected by subsidies. It is interesting to note that the three most widely grown crops in the study area are also the only three crops for which subsidies are observed in the baseline data set. These crops are maize, cotton, and cereals. As the price of water is increased, both approaches predict that the overwhelming majority of croplands in the case study basin will be converted to cereals, which is the only dryland crop included in the study. It would be interesting to see if the PMP approach would predict that the same amount of land would be converted to cereals production if the profitability of cereals was not enhanced by the presence of a subsidy.

Although the PMP approach has the capacity to predict that capital can be substituted for water, this is not observed at any water use location in the case study area. It seems reasonable that irrigation water users might invest in water-saving irrigation technology or other factors of production in response to water price increases. In an analysis of irrigation data from the Maipo River basin in Chile, Cai et al. (2008) found that irrigation investment, machinery, and labor can all be substitutes for water. The reason that substitution is not observed in this study may be because all inputs apart from land and water are aggregated into a single input, “capital”. This aggregated input includes both inputs that are substitutes for water and other inputs that are not substitutes. However, it is not clear whether disaggregating this input into its component parts would be feasible given the constant elasticity of substitution production function assumed here. This function uses a single elasticity value to estimate substitution between all inputs, which may not be appropriate if more inputs are added to the production function.

It should not be surprising that the two pricing policies compared in the third investigation estimate similar water use and welfare impacts. The policies are essentially the same except that the method used to price groundwater use is different under the second policy. Both policies are modeled using the same environmental and groundwater sustainability constraints. Both policies equate marginal values of water use throughout the case study area, although the second policy does not equate marginal values exactly. Because water users are the same under each of the water pricing policies, equating marginal values of water use throughout the study are should maximize net benefits, with the same net benefit total observed under both policies.

However, it can also be seen that the distribution of benefits under the first policy is different from the distribution observed under the second policy. Both policies predict significant impacts to the agriculture sector. Both policies also predict that these impacts will be concentrated on low value crops such as maize, cotton, and fodder. However, the second policy predicts that it will profitable to continue growing these crops at reduced levels while the first policy predicts that these crops will go out of production entirely. It is frequently claimed that low value crops will not be profitable if water for irrigation is priced volumetrically (e.g., Gomez and Limon, 2004; Latinopoulos, 2008), a conclusion that is supported by this study. If volumetric pricing is to be introduced, this study suggests that using an energy price to control groundwater use will reduce impacts on growers of low value crops.

On the other hand, the second pricing policy increases impacts on water users that do not have access to groundwater supplies. Because more groundwater is used under the second policy, the surface water price must be increased in order to reduce surface water use and compensate for reduced groundwater base flow. This means that irrigation water use locations without access to groundwater face higher prices under the second policy.

The second pricing policy also reduces benefits to large urban/domestic users while increasing benefits to small users. Because drawdown increases with higher pumping rates, the marginal cost of water use increases with water use under the second policy. The result is that large water users have higher marginal costs than smaller users. This is seen in the retail prices presented in Table 4.4, which are a function of marginal supply costs. Table 4.4 also shows that retail prices for smaller urban/domestic locations are lower under the second pricing policy than under the first policy.

6 Conclusions

In the first investigation, a hydro-economic modelling approach is used to investigate the potential impacts of a water pricing policy that meets the objectives of the EU WFD. A volumetric water pricing policy is simulated in which all wholesale water users in a river basin are charged the same price. A major conclusion is that water prices would have significant economic impacts on the agriculture sector. These impacts appear to be concentrated on growers of low value crops such as maize, cotton, and fodder crops, which would be unprofitable to grow even at lower water prices.

Because impacts on the agriculture sector are more significant than impacts on other sectors, two approaches to modelling the economic behavior of farmers are compared in a second investigation. The PMP and residual imputation approaches predict similar changes in irrigation water use as a function of water price changes. Although the PMP approach has the capacity to predict a much wider variety of responses to water price changes, these responses are not observed. The PMP approach predicts that deficit irrigation will be a profit-maximizing strategy for many high-value crops, but does not predict that irrigated areas cultivated with low-value crops in the baseline data set will be converted to high-value crops. Because the proportion of irrigated areas cultivated with high-value crops in the baseline data set is small, the result is that land and water use levels predicted by the two approaches are essentially the same.

The prediction that high-value irrigated crops will not replace low-value irrigated crops is not unreasonable given behavior observed in the baseline data set. This highlights the limitations of using economic models calibrated to observed behavior to predict responses to new conditions. The introduction of new water pricing policies would be a significant change for the irrigation economy in the case study basin. In this case, past behavior is not necessarily a guide to future decision-making. If volumetric pricing is to be introduced, either as a tool to achieve environmental quality goals or to achieve economic efficiency objectives, then the results of this study suggest these efforts should be accompanied by agronomic and irrigation water management initiatives to help farmers succeed in this new economic environment.

In a third investigation, two pricing policies that meet the water pricing objectives of the WFD are compared. The second pricing policy, in which the energy price is used as a surrogate for a groundwater price, shifts a small portion of costs imposed by higher water prices from low value crops to high value crops and from small urban/domestic locations to larger locations. Because growers of low value crops will suffer the most from water price increases, the second policy offers the advantage of reducing this burden. In addition, because of difficulties associated with monitoring groundwater use, the second policy may be easier to implement in practice.

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8 Papers

The following papers are included in this thesis:

I Riegels, N., Jensen, R., Bensasson, L., Banou, S., Møller, F., Bauer-Gottwein, P., 2011. Estimating resource costs of compliance with EU WFD ecological status requirements at the river basin scale. Journal of Hydrology, 396 (197-214).

II Riegels, N., Sturm, V., Doulgeris, C., Jensen, R., Møller, F., Bauer-Gottwein, P., 2011. Comparison of two approaches for predicting farmer responses to water price changes in a hydro-economic modelling study. Submitted to Water Resources Research.

III Riegels, N., Pulido Velazquez, M., Sturm, V., Doulgeris, C., Jensen, R., Møller, F., Bauer-Gottwein, P., 2011. Comparison of two water pricing policies in a hydro-economic modelling study. Submitted to the Journal of Water Resources Planning and Management (ASCE).

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